

# 16th Annual Report 2007

**UNECE Convention on Long-range  
Transboundary Air Pollution**

**International Cooperative Programme  
on Integrated Monitoring of Air Pollution  
Effects on Ecosystems**

**Sirpa Kleemola and Martin Forsius**

ENVIRONMENTAL  
PROTECTION







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**Convention on Long-range Transboundary Air Pollution**

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Integrated Monitoring of Air Pollution Effects on Ecosystems**

**Sirpa Kleemola and Martin Forsius (eds.)**

**wge** Working Group on Effects of the  
Convention on Long-range  
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# Summary

## Background and objectives of ICP IM

Integrated monitoring of ecosystems means physical, chemical and biological measurements over time of different ecosystem compartments simultaneously at the same location. In practice, monitoring is divided into a number of compartmental subprogrammes which are linked by the use of the same parameters (cross-media flux approach) and/or same or close stations (cause-effect approach).

The International Cooperative Programme on Integrated Monitoring of Air Pollution Effects on Ecosystems (ICP IM) is part of the Effects Monitoring Strategy under the Convention on Long-range Transboundary Air Pollution (LRTAP). The main objectives of the ICP IM are:

- To monitor the biological, chemical and physical state of ecosystems (catchments/ plots) over time in order to provide an explanation of changes in terms of causative environmental factors, including natural changes, air pollution and climate change, with the aim to provide a scientific basis for emission control.
- To develop and validate models for the simulation of ecosystem responses and use them (a) to estimate responses to actual or predicted changes in pollution stress, and (b) in concert with survey data to make regional assessments.
- To carry out biomonitoring to detect natural changes, in particular to assess effects of air pollutants and climate change.

The full implementation of the ICP IM will allow ecological effects of heavy metals, persistent organic substances and tropospheric ozone to be determined. A primary concern is the provision of scientific and statistically reliable data that can be used in modelling and decision making.

The ICP IM sites (mostly forested catchments) are located in undisturbed areas, such as natural parks or comparable areas. The ICP IM network presently covers forty-seven sites from seventeen countries. The international Programme Centre is located at the Finnish Environment Institute in Helsinki. The present status of the monitoring activities is described in detail in Section 1 of this report.

A manual detailing the protocols for monitoring each of the necessary physical, chemical and biological parameters is applied throughout the programme (Manual for Integrated Monitoring 1998).

## Recent assessment activities within the ICP IM

Assessment of data collected in the ICP IM framework is carried out at both national and international levels. Key recent tasks regarding international ICP IM data have been:

- Input-output and proton budgets
- Trend analysis of bulk and throughfall deposition and runoff water chemistry
- Assessment of biological data using multivariate gradient analysis
- Dynamic modelling and assessment of the effects of different emission / deposition scenarios, including confounding effects of climate change processes
- Assessment of concentrations, pools and fluxes of heavy metals
- Empirical thresholds for N deposition (soil C/N ratios, input-output budgets)
- Compilation of available information on cause-effect relationships of forest ecosystems



## Conclusions from recent international studies

### Input-output and proton budgets, C/N interactions

Ion mass budgets have proved to be useful for evaluating the importance of various biogeochemical processes that regulate the buffering properties in ecosystems. Long-term monitoring of mass balances and ion ratios in catchments/plots can also serve as an early warning system to identify the ecological effects of different anthropogenically derived pollutants, and to verify the effects of emission reductions.

The first results of input-output and proton budget calculations were presented in the 4th Annual Synoptic Report (ICP IM Programme Centre 1995) and the updated results regarding the effects of N deposition were presented in Forsius *et al.* (1996). Data from selected ICP IM sites have also been included in European studies for evaluating soil organic horizon C/N-ratio as an indicator of nitrate leaching (Dise *et al.* 1998, MacDonald *et al.* 2002). Soil water fluxes for budget calculations have been estimated using a water balance model (Starr 1999). New results regarding the calculation of fluxes and trends of S and N compounds were presented in a scientific paper prepared for the Acid Rain Conference, Japan, December 2000 (Forsius *et al.* 2001). A scientific paper regarding calculations of proton budgets was published in 2005 (Forsius *et al.* 2005).

The budget calculations showed that there was a large difference between the sites regarding the relative importance of the various processes involved in the transfer of acidity. These differences reflected both the gradients in deposition inputs and the differences in site characteristics. The proton budget calculations showed a clear relationship between the net acidifying effect of nitrogen processes and the amount of N deposition. When the deposition increases also N processes become increasingly important as net sources of acidity.

A critical deposition threshold of about 8-10 kg N ha<sup>-1</sup> a<sup>-1</sup>, indicated by several previous assessments, was confirmed by the input-output calculations with the ICP IM data (Forsius *et al.* 2001). The output flux of nitrogen was strongly correlated with key ecosystem variables like N deposition, N concentration in organic matter and current year needles, and N flux in litterfall (Forsius *et al.* 1996). Soil organic horizon C/N-ratio seems to give a reasonable estimate of the annual export flux of N for European forested sites receiving throughfall deposition of N up to about 30 kg N ha<sup>-1</sup> a<sup>-1</sup>. When stratifying data based on C/N ratios less than or equal to 25 and greater than 25, highly significant relationships were observed between N input and nitrate leached (Dise *et al.* 1998, MacDonald *et al.* 2002, Gundersen *et al.* 2006). Such statistical relationships from intensively studied sites can be efficiently used in conjugation with regional monitoring data (e.g. ICP Forests and ICP Waters data) in order to link process level data with regional-scale questions.

Sulphur budgets calculations indicated a net release of S from many ICP IM sites, indicating that the soils are releasing previously accumulated S. Similar results have been obtained in other recent European plot and catchment studies.

The reduction in deposition of S and N compounds at the ICP IM sites, caused by the 'Protocol to Abate Acidification, Eutrophication and Ground-level Ozone' of the LRTAP Convention ('Gothenburg protocol'), was estimated for the year 2010 using transfer matrices and official emissions. Implementation of the protocol will further decrease the deposition of S and N at the ICP IM sites in western and north western parts of Europe, but in more eastern parts the decrease will be smaller (Forsius *et al.* 2001).

Results from the ICP IM sites have also been summarised in an assessment report prepared by the Working Group on Effects of the LRTAP Convention (Sliggers and Kakebeeke 2004, Working Group on Effects 2004).

It should also be recognized that there are important links between N deposition and the sequestration of C in the ecosystems (and thus direct links to climate change processes). These questions were studied in the CNTER-project in which data from both the ICP IM and EU/Intensive Monitoring sites were used (Gundersen *et al.* 2006).

## Trend analysis

Empirical evidence on the development of environmental effects is of central importance for the assessment of success of international emission reduction policy. First results from a trend analysis of monthly ICP IM data on bulk and throughfall deposition as well as runoff water chemistry were presented in Vuorenmaa (1997). ICP IM data on water chemistry have also been used for a trend analysis carried out by the ICP Waters and presented in the Nine Year Report of that programme (Lükewille *et al.* 1997).

Calculations on the trends of N and S compounds, base cations and hydrogen ions were made for 22 ICP IM sites with available data across Europe (Forsius *et al.* 2001). The site-specific trends were calculated for deposition and runoff water fluxes using monthly data and non-parametric methods.

Statistically significant downward trends of  $\text{SO}_4$ ,  $\text{NO}_3$  and  $\text{NH}_4$  bulk deposition (fluxes or concentrations) were observed at 50% of the ICP IM sites. Sites with higher N deposition and lower C/N-ratios clearly showed higher N output fluxes, and the results were consistent with previous observations from European forested ecosystems. Decreasing  $\text{SO}_4$  and base cation trends in runoff waters were commonly observed at the ICP IM sites. At some sites in the Nordic countries decreasing  $\text{NO}_3$  and  $\text{H}^+$  trends (increasing pH) were also observed. The results partly confirm the effective implementation of emission reduction policy in Europe. However, clear responses were not observed at all sites, showing that recovery at many sensitive sites can be slow and that the response at individual sites may vary greatly.

Data from ICP IM sites were also used in a study of the long-term changes and recovery at nine calibrated catchments in Norway, Sweden and Finland (Moldan *et al.* 2001, RECOVER:2010 project). Runoff responses to the decreasing deposition trends were rapid and clear at the nine catchments. Trends at all catchments showed the same general picture as from small lakes in Scandinavia.

It was agreed at the ICP IM Task Force meeting in 2004 that a new trend analysis should be carried out. The preliminary results were presented in Kleemola (2005) and the updated results in the 15<sup>th</sup> Annual Report (Kleemola *et al.* 2006). Statistically significant decreases in  $\text{SO}_4$  concentrations were observed at a majority of sites in both deposition and runoff/soil water quality. Increases in ANC (acid neutralising capacity) were also commonly observed. For  $\text{NO}_3$  the situation was more complex, with fewer decreasing trends in deposition and even some increasing trends in runoff/soil water.

## Assessment of biological data using multivariate gradient analysis

The effect of pollutant deposition on natural vegetation, including both trees and understorey vegetation, is one of the central concerns in the impact assessment and prediction. The first assessment of vegetation monitoring data at ICP IM sites with regards to N and S deposition was carried out by Liu (1996). Vegetation monitoring was found useful in reflecting the effects of atmospheric deposition and soil water

chemistry, especially regarding sulphur and nitrogen. The results suggested that plants respond to N deposition more directly than to S deposition with respect to vegetation indices.

De Zwart (1998) carried out an exploratory multivariate statistical gradient analysis of possible causes underlying the aspect of forest damage at ICP IM sites. These results suggested that coniferous defoliation, discolouration and lifespan of needles in the diverse phenomena of forest damage are for respectively 18%, 42% and 55% explained by the combined action of ozone and acidifying sulphur and nitrogen compounds in air.

From the previous ordination exercises it was concluded that the applied statistical techniques are capable of revealing underlying structure and possible cause-effect relationships in complex ecological data, provided that analysed gradients have an adequate range to be interpolated. Since the data obtained were unexpectedly poor in the span of environmental gradients, the results of the presented statistical ordination only indicated correlative cause-effect relationships with a limited validity. The poor span of gradients could be attributed to the relative scarcity of biological effect data and the occurrence of missing observations both in the chemical and biological data sets. It was concluded, that the power of the vegetation monitoring in impact assessment would increase considerably with improvements in the ICP IM data reporting and inclusion of additional sites.

As a separate exercise, the epiphytic lichen flora of 25 European ICP IM monitoring sites, all situated in areas remote from local air pollution sources, was statistically related to measured levels of SO<sub>2</sub> in air, NH<sub>4</sub><sup>+</sup>, NO<sub>3</sub><sup>-</sup> and SO<sub>4</sub><sup>2-</sup> in precipitation, annual bulk precipitation, and annual average temperature (van Herk *et al.* 2003, de Zwart *et al.* 2003). It was concluded that long distance transport of nitrogen air pollution is important in determining the occurrence of acidophytic lichen species, and constitutes a threat to natural populations that is strongly underestimated so far.

### Dynamic modelling and assessment of the effects of emission/deposition scenarios

In a policy-oriented framework, dynamic models are needed to explore the temporal aspect of ecosystem protection and recovery. The critical load concept, used for defining the environmental protection levels, does not reveal the time scales of recovery. Dynamic models have been developed and used for the emission/deposition scenario assessment at selected ICP IM sites (e.g. Forsius *et al.* 1997, 1998a 1998b, Posch *et al.* 1997, Jenkins *et al.* 2003). These models are flexible and can be adjusted for the assessment of alternative scenarios of policy importance.

These modelling studies have shown, that the recovery of soil and water quality of the ecosystems is determined by both the amount and the time of implementation of emission reductions. According to the models, the timing of emission reductions determines the state of recovery over a short time scale (up to 30 years). The quicker the target level of reductions is achieved, the more rapidly the surface water and soil status recover. For the long-term response (> 30 years), the magnitude of emission reductions is more important than the timing of the reduction. The model simulations also indicate that N emission controls are very important to enable the maximum recovery in response to S emission reductions. Increased nitrogen leaching has the potential to not only offset the recovery predicted in response to S emission reductions but further to promote substantial deterioration in pH status of freshwaters and other N pollution problems in some areas of Europe.

At the 17th session of the Executive Body of the Convention in December 1999 the importance of the monitoring and dynamic modelling of recovery was underlined. ICP IM participates in a joint coordinated exercise on dynamic modelling together



with other ICPs. UK is leading this modelling work in ICP IM. The work has strong links to projects financed by the Nordic Council of Ministers and the EU. Priority in the ICP IM work is given to site-specific modelling. The role of ICP IM in this activity is to provide detailed and consistent physical and chemical data and long time-series of observation for key sites against which model performance can be assessed and key uncertainties identified (see Jenkins *et al.* 2003).

Work is also on-going to predict potential climate change impacts on air pollution related processes at these sites. The large EU-project EURO-LIMPACS ([www.eurolimpacs.ucl.ac.uk](http://www.eurolimpacs.ucl.ac.uk), 2004-2009) is studying the global change impacts on freshwater ecosystems. The institutes involved in the project are using data collected at ICP IM and ICP Waters sites as key datasets for the modelling, time-series and experimental work of the project. A first modelling assessment on the global change impacts on acidification recovery has been carried out in the project (Wright *et al.* 2006). The results showed that climate/global change induced changes may clearly have a large impact on future acidification recovery patterns, and need to be addressed if reliable future predictions are wanted (decadal time scale). However, the relative significance of the different scenarios was to a large extent determined by site-specific characteristics. For example, changes in sea-salt deposition were only important at coastal sites and changes in decomposition of organic matter at sites which are already nitrogen saturated.

## **Pools and fluxes of heavy metals**

The work to assess concentrations, stores and fluxes of heavy metals at ICP IM is led by Sweden. Preliminary results on concentrations, fluxes and catchment retention have been reported to the Working Group on Effects (document EB.AIR/WG.1/2001/10). Considerable retention of Cd, Cu, Ni, Pb and Zn (80-95 % of total input) was observed at some sites with available detailed information. A scientific paper on the results will be finalised in 2007. The main findings on heavy metals budgets and critical loads at ICP IM sites were presented in the 15<sup>th</sup> Annual Report (Bringmark *et al.* 2006). In many national studies on ICP IM sites, detailed site-specific budget calculations of heavy metals (including mercury) have improved the scientific understanding of ecosystem processes, retention times and critical thresholds. ICP IM sites are also used for dynamic model development of these compounds.

## **Compilation of available information on cause-effect relationships of forest ecosystems**

A report summarising available information from the ICP Forests and ICP IM programmes on cause-effect relationships of forest ecosystems has been prepared (de Vries *et al.* 2002). The results were also officially reported to the Working Group on Effects in 2002 (EB.AIR/WG.1/2002/15).

## **Planned activities**

- Maintenance and development of a central ICP IM database at the Programme Centre.
- Continued assessment of the long-term effects of air pollutants to support the implementation of emission reduction protocols, including:
  - Assessment of trends.
  - Calculation of ecosystem budgets, empirical deposition thresholds and site-specific critical loads.
  - Dynamic modelling and scenario assessment.
  - Comparison of calculated critical load exceedances with observed ecosystem effects.

- Calculation of pools and fluxes of heavy metals at selected sites.
- Assessment of cause-effect relationships for biological data, particularly vegetation.
- Coordination of work and cooperation with other ICPs, particularly regarding dynamic modelling (all ICPs), cause-effect relationships in terrestrial systems (ICP Forests, ICP Vegetation), and surface waters (ICP Waters).
- Cooperation with external organisations and programmes, particularly Global Terrestrial Observing System (GTOS, <http://www.fao.org/gtos/index.html>) and International Long Term Ecological Research Network (ILTER, <http://www.ilternet.edu/>).
- Participation in projects with a global change perspective. Data from sites in the ICP IM network are currently used in the EU-projects 'Integrated project to evaluate impacts of global change on European freshwater ecosystems (EURO-LIMPACS, <http://www.eurolimpacs.ucl.ac.uk/>)', and 'A long-term Biodiversity, Ecosystem and Awareness Research Network (ALTER-Net, <http://www.alter-net.info/default.asp>)'.
- Initiation of new assessment activities regarding global change impacts (e.g. Parr *et al.* 2002).

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# 1 ICP IM activities, monitoring sites and available data

## 1.1

### **Review of the ICP IM activities in 2006-2007**

#### **Meetings**

- Lars Lundin represented the ICP IM programme at the 22<sup>nd</sup> ICP Forests TF in Tallinn, Estonia, 20-24 May 2006.
- ICP IM programme was represented by Martin Forsius at the BIOGEOMON conference in San Francisco, USA, 26-30 June, 2006
- Lars Lundin and Martin Forsius participated in the Extended Bureau of Working Group on Effects (WGE) meeting on 30 August, 2006 in Geneva as well as the WGE meeting 31 August - 1 September. ICP IM activities and the Annual Report 2006 were presented.
- Martin Forsius took part in the EC Climate Change workshop held in Brussels, Belgium, 25-26 September 2006.
- Workshop on confounding factors in recovery from acid deposition in surface waters was organised by ICP Waters in collaboration with ICP IM 9-10 October 2006 in Bergen, Norway. The workshop was held back-to-back with the 22<sup>nd</sup> ICP Waters Task Force meeting (11-12 October 2006), where Martin Forsius represented the ICP IM Programme Centre.
- Martin Forsius took part in the EU / EURO-LIMPACS meeting held in Amsterdam, the Netherlands, 30-31 October 2006.
- Lars Lundin participated in the Extended Bureau meeting of the Working Group on Effects Extended Bureau meeting in Geneva on the 14-16 February 2007. Key issues discussed were related to the review process and potential revision of the protocols.
- Lars Lundin and Martin Forsius took part in the EU / Alter-Net meeting, 6-9 February 2007 in Mallorca, Spain.
- Martin Forsius represented the ICP IM programme at the Life-Watch meeting 13-14 March 2007 in Stuttgart, Germany.
- Martin Forsius participated in the EU / EURO-LIMPACS meeting in Leibzig, Germany, 16-20 April, 2007
- The fifteenth meeting of the Programme Task Force on ICP Integrated Monitoring was organised in Grafenau, Germany, 10 May, 2007. A one-day workshop on the assessment of ICP IM data was held prior to the Task Force meeting on 9 May.

## Projects, data issues

- Data from both ICP IM and the EU/ICP Forests Intensive Monitoring Programme has been used in the EU-project CINTER (Carbon and nitrogen interactions in forest ecosystems, [www.flec.kvl.dk/cnter](http://www.flec.kvl.dk/cnter)). The final report of the project was published in 2006 (Gundersen *et al.* 2006). The project was of strategic importance because it allowed the use of ICP IM data in relation to global change issues (C-sequestration).
- Data from sites in the ICP IM network are also used in the EU-projects 'Integrated project to evaluate impacts of global change on European freshwater ecosystems (EURO-LIMPACS, [www.eurolimpacs.ucl.ac.uk](http://www.eurolimpacs.ucl.ac.uk))', and 'A long-term biodiversity, ecosystem and awareness research network (ALTER-Net, [www.alter-net.info](http://www.alter-net.info))'.
- After December 1<sup>st</sup> 2006 the National Focal Points (NFPs) reported their 2005 results to the IM Programme Centre. The Programme Centre carried out standard check up of the results and incorporated them into the IM database.
- Laboratories participating in the ICP IM Programme took part in the intercomparison tests organised by ICP Waters and EMEP.

## Scientific work in priority topics

Scientific work regarding four priority topics has continued:

- Calculation of pools and fluxes of heavy metals and relations to critical limits and risk assessment (led by Sweden). An official document was produced for the WGE meeting in 2006 on main findings on heavy metal budgets and critical loads at ICP IM sites. Results were also presented at the Task Force meeting in Riga in 2006. A scientific paper will be finalised in 2007.
- Dynamic modelling (led by CEH in UK in cooperation with the Programme Centre and NIVA, Norway). This work has strong links to projects financed by the EU. ICP IM participates in a joint coordinated exercise on dynamic modelling together with other ICPs (Joint Expert Group on Dynamic Modelling, JEG DM). Priority in the ICP IM work is given to site-specific modelling activities. A scientific paper based on the first results from site-specific dynamic modelling on climate change impacts on acidification recovery (with ICP Waters, based on EURO-LIMPACS results) has been prepared (Wright *et al.* 2006). An official document on the first results was presented to the WGE in 2005. The use of dynamic modelling forecasts to derive future target loads for N and S in atmospheric deposition, including climate change impacts, is presented in this report (M. Hutchins).
- Assessment of cause-effects relationships for biological data, particularly vegetation. The new contact appointed by ICP IM Task Force in 2004 is the NFP of Italy in collaboration with the NFP of Austria (in collaboration with ICP Forests Intensive Monitoring). This work is closely connected to the ALTER-Net project.
- Calculation of fluxes and trends of N and S compounds, base cations and acidity (led by the Programme Centre). Priority is given to calculation of proton budgets, N processes and budgets and C/N interactions. Work on proton and N budgets was published in 2005 (Dise *et al.* 2005, Forsius *et al.* 2005). Updated results from the trend analyses on observed concentrations/fluxes were presented at the Task Force meeting 2006 and in the Annual Report 2006. The final report of the CINTER-project was published in 2006 (Gundersen *et al.* 2006). A summary based on the CINTER-project results will be presented at the WGE 2007.



## Reports

ICP IM will produce the following reports to the meeting of Working Group on Effects September 2007.

- 16<sup>th</sup> ICP IM Annual Report 2007
- Contribution to Joint Report of the ICPs
- Summary report on C/N -interactions and nitrogen effects in European forest ecosystems.
- Report on support of effect-based approaches for the review and possible revision of the Convention protocols (together with ICP Waters).

### 1.2

## Activities and tasks planned for 2007-2008

Activities/tasks related to the programme's present objectives, carried out in close collaboration with other ICPs/ Task Force

- Maintenance and development of central ICP IM database at the Programme Centre.
- Finalisation of a scientific paper on heavy metals (2007)
- Finalisation of summary report on C/N -interactions and nitrogen effects in European forest ecosystems (2007).
- Finalisation of report on support of effect-based approaches for the review and possible revision of the Convention protocols (2007, together with ICP Waters).
- Interim report on site specific dynamic modelling for acidification related to climate at ICP IM sites (2008).
- Report on links between climate change and air pollution effects using site-specific data (2008).
- Progress report on biodiversity issues (2008, in collaboration with EU ALTER-Net project).
- Arrangement of the 16<sup>th</sup> Task Force meeting (2008).
- Preparation of the 17<sup>th</sup> ICP IM annual report (2008).
- Participation in meetings of the WGE, other ICPs and the JEG DM.

## Activities/tasks aimed at further development of the programme

- Participation in the activities of external organisations, particularly Global Terrestrial Observing System (GTOS) and the International Long Term Ecological Research Network (ILTER).
- Participation in the EU-projects EURO-LIMPACS and ALTER-Net.

## Published reports and articles 2006-2007

### Evaluations of international ICP IM data and related publications

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#### 1.4

### Monitoring sites and data

The following seventeen countries have continued data submission to the ICP IM data base during the 2000 - 2006 period: Austria, Belarus, Canada, Czech Republic, Denmark, Estonia, Finland, Germany, Iceland, Italy, Latvia, Lithuania, Norway, Portugal, Russian Federation, Sweden, and United Kingdom. Denmark has discontinued monitoring, but has reported data from year 2002.

Presently the number of ICP IM sites with on-going data submission, data for at least part of the period 2000 – 2006, is forty-seven, most of the sites are European. An overview of the data reported internationally to the ICP IM database is given in Table 1.1. Additional earlier reported data are available from sites outside those presented in Table 1.1 and Figure 1.1. These sites have either been suspended or taken out of the IM network and used for regional monitoring. Location of the IM monitoring sites with data from recent years are shown in Figure 1.1, including some sites without data from period 2000 - 2006.



Figure 1.1 Geographical location of ICP IM sites with data from recent years



Table I.1 Internationally reported data from ICP IM sites (- subprogramme not possible to carry out, \* or forest health parameters in former Forest stands/Trees).

AREA	SUBPROGRAMME				TF	SF	SC	SW	GW	RW	LC	FC	LF	RB	LB	*	FD	VG	BI	VS	EP	AL	MB	BB	BV	
	AM	AC	PC	MC																						
		meteorol.	air chemistry	precipit. chemistry	moss chemistry	throughfall	stemflow	soil chemistry	soil water chemistry	ground-water chemistry	runoff water c.	lake water chemistry	foliage chemistry	litterfall chemistry	hydrobiol. of streams	hydrobiol. of lakes		forest damage	vegetation	bio-elements	vegetation structure	trunk epiphytes	aerial gr. algae	microbial. decompos.	bird inventory	vegetation inventory
AT01	95-05	95-05	93-05			93-05	98-04	95-98	03-05		95-05	-							93-99	93		93-98				
BY02	89-05	89-05	89-05					95-98			95-05															
CA01	88-03		88-03							88-03	88-03															
CZ01	89-05	89-05	89-05	89		89-05					89-05															
CZ02			00-05			00-05					00-05															
DE01	90-05	90-05	90-05	90		90-05	90-05	90	90-05	88-05	90-05	-	90-05	90-05			90-05	90-01			00	92-95		94-05	91-02	
DE02	67-05	98-05	98-05			98-05		04	98-05	98-05		98-05	06													
DK01			92-02			92		86	92-02		-				-											
DK02			97-02								97-02															
DK03			94-00			94-00		95	94-00		-							95								
EE01	95-05	94-05	94-05	94-05	94-05	94-05	94-05	94-05	94-05	95-96	-	-	94-05	94-05	-	-	94-95	94-97				94-04	94-02		94	
EE02	94-05	98-05	94-05	94-05	94-05	94-05	94-05	94-05	95-05	95-01	94-05	96	94-02	94-05			96-05	96				94-01	96-00	98		
FI01	88-05	94-05	88-05	88-96	88-96	89-02	89-99	88-89	89-01	88-05	88-05	87-05	88-01	90-97		90-93	88-91	88-98				88-97	90	87-89	87	
FI03	88-05	93-00	88-05	89-96	89-96	89-02	89-99	88	89-01	88-05	88-05	87-05	88-01	90-97		90	88-91	90-98				90-97	90-91	87-89		
FI04	88-05	89-05	88-05	89-96	89-96	89-02	89-97	89	89-01	88-05	88-05	86-05	89-01	90-97			89-91	89-98				89-98	90-91	87-89		
FI05	88-05		88-05	91-96	91-96	89-97	89-97	88	89-96		89-05	87-05	88-01	90-97			88-91	89-98				89-97	90-91	88-89		
GB01	88-05	91-05	88-05					90		90-91	88-05	-														
GB02	88-05	91-05	88-05			88-91	88-91		90-91		88-05	-													96	
IS01			97-04							98-04	97-04															
IT01	93-05	93-05	93-05			93-05	93-05	93-00	93-05		00-05	-	93-05	00	-	-	92-05			05	92	93-03				
IT02	77-05	93	93-05			93-05	93-05	93-00	93-05		-	-	93-00	00	-	-	92-05				92	93-03				
IT03	92-05	93-05	92-05			94-05	94-00	93-95	95-00		02-05	-	93-05	94	-	-	93-05	95			99-05	92				
IT04	92-00	93-97	92-00			94-00	94-00	93-95	95-00		-	-	93-95	94	-	-	93-00				99-05	92				
IT05	97-05	97-05	97-01			97-01	97-01	95	02-05		-	-	97-05		-	-	97-05				99-05					
IT06	99-05	97-05	97-05			97-05	97-05	95		-	-	-	97-05		-	-	97-05				99-05					
IT07	97-05	97-05	97-05			97-05	97-00	95		98	98	-	97-05		-	-	97-05				99-05					
IT08	98-05	97-05	97-05			97-05	97-05	95		97-05	97-05	-	97-05		-	-	97-05				99-05					
IT09	97-05	97-05	97-05			97-05	97-99	95	02-05		97-05	-	97-05		-	-	97-05				99-05					
IT10	97-05	00-05	97-05			97-05		95	05		-	-	97-05		-	-	97-05				99-05					
IT11		97-05	97-99			97-99		95			-	-	97-05		-	-	97-05				99-05					
IT12	97-01	97-05	97-05			97-05	97-00	95		-	-	-	97-05		-	-	97-05				99-05					
IT13	97-05	97-05						95			-	-	97-05		-	-	97-05				99-05					
LT01	93-04	93-04	93-04	93		93-04		93-00	94-04	93-04	93-04			99-04			00-04	93-04		02	93-02	93-01		93		
LT03	95-04	95-04	95-04			95-04		94-00	95-04	95-04	95-04			99-04	95-98		00-04	94-04		02	94-02	94-01		94		
LV01	93-05	93-05	93-05	94-98		94-05	94-05	94-03	94-05	94-05	93-05	-	94-05	94-05	95-98	-	94-05	94-03				94-03	01-05	96-98		
LV02	93-05	94-05	93-05	94-98		94-05	94-05	94-03	94-05	94-05	93-05	93-98	94-05	94-05	95-98	95-98	94-05	94-03		01	94-03	01-05	96-98			
NO01	87-05	87-05	87-05	92		89-05	86	89-05	89-05	87-88	87-05	-	86			-	91-03	86			86					
NO02	87-91	87-05	87-05	88		89-05	89	89-05		87-05	87-05	-	89			-	92-03	89								
PT01	88-01	89-01	94-01								90-01															
RU03	89-94	89-05	89-98																							
RU04	89-05	89-05	89-98	90											93-99		93-05	93-02			93		94-96			
RU12	93-94	93-05	93-94																							
RU14	94	94-05	94-98																							
SE04	87-05	88-05	87-05	95		87-05	95		87-05	79-05	87-05	-	99-05	96-04	-	-	97-01	95-04				96-01	92-05	95-05		
SE14	96-05	96-05	96-05	95		96-05	96-05	95	95-05	96-05	96-05	-	99-04	95-05	-	-	97-01	82-04				97	97-05	95-05		
SE15	97-05	96-05	96-05			96-05	97		95-05	97-05	96-05	-	97-05	95-05	-	-	98-01	96-04				98	97-05	95-05		
SE16	99-05	99-05	99-05			99-05			00-05	00-05	99-05		99-05	00-05			00-01	99-04		99-04	00	00-05	00-05			

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## 2 Effects based approaches for S and N

*Summary prepared by the Programme Centres of ICP IM and ICP Waters for the Gothenburg Protocol review process*

### **Abstract**

Monitoring data from ICP Waters and ICP IM as well as other sources show clear and large regional trends in deposition and surface water chemistry in response to the large decreases in deposition of sulphur since the mid-1980s. Waters have become less acidic and less toxic to biota. At many sites sulphate concentrations now approach new levels expected following full implementation of the Gothenburg protocol. Nitrate, on the other hand, does not show consistent trends, and most sites are far from steady-state conditions. Dynamic models indicate that a significant number of sites in several regions of Europe will continue to be acidified after 2010. Biological recovery has begun in many regions, but lags behind chemical recovery. Future climate change will affect acidification and recovery.

#### 2.1

### **Assessment of trends**

The strongest evidence that emissions control programs are having their intended effect comes from a consistent pattern of recovery (decreasing sulphate and increasing pH and alkalinity) across a large number of surface water sites. The most recent evaluation of trends in ICP Waters data consists of chemical records from the period 1994 to 2004 for 174 sites (68 from Europe, 106 from North America) grouped in twelve fairly homogeneous regions with regard to deposition level and acid-sensitivity (de Wit and Skjelkvåle, in preparation). The most important finding is the widespread chemical recovery in streams and lakes in most regions in Europe and North America. This was largely due to the decline in sulphate. All regions except two showed a significant increase in pH and/or alkalinity, and/or acid neutralizing capacity. The regions without signs of chemical recovery were Ontario and the Virginia Blue Ridge mountains in North America. No universal rise or decline in nitrate was detected, and nitrate trends varied considerably within each region. Deposition of N in a subset of the sites did not show a strong decline, and in very few sites similar trends in N deposition and N runoff were found. The monitoring data also indicate that biological recovery (fish, invertebrates) has begun in many regions, but lags behind chemical recovery.

Data from ICP IM sites have also been used in trend assessments for the period 1993-2003 (Kleemola and Forsius 2006). This analysis confirmed the previously observed regional-scale decreasing trends of S in deposition and runoff/soil water. Acid-sensitive ICP IM sites in northern Europe showed recovery from acidification. The situation regarding N was quite different with few statistically significant decreasing trends of N in deposition and both decreasing and increasing trends of nitrate in runoff/soil

water. Site-specific characteristics are important in determining the response to N (and S) emission reductions. It has also been concluded that long distance transport of nitrogen air pollution is important in determining the occurrence of acidophilic lichen species at ICP IM sites, and constitutes a threat to natural populations that has been underestimated so far (van Herk *et al.* 2003, de Zwart *et al.* 2003). The N issue thus clearly requires continued attention as a European air pollution problem.

Environmental factors other than deposition are expected to affect chemical and biological recovery of freshwaters and soils in response to reduced deposition inputs. Climate change may both enhance and delay recovery depending on region and variable considered (see modelling section below). Dissolved organic carbon (DOC) is of great interest in analysis of surface water recovery, because it is an indicator of organic (natural) acidity which may counteract the positive effect of declining sulphate. For the period 1990-2004, a widespread increase in DOC was found in formerly glaciated parts of North America and Europe. This increase is likely related to changes in both deposition loads and climatic factors.

## 2.2

### **Ion mass and proton budget calculations, C/N interactions**

Long-term monitoring of mass balances and ion ratios in catchments/plots verifies the impacts of emission reductions and serves as an early warning system to identify the ecological effects of different anthropogenic pollutants. Proton budget calculations showed that there was a large difference between the ICP IM sites regarding the relative importance of the various processes involved in the transfer of acidity (Forsius *et al.* 2005). These differences reflected both the gradients in deposition inputs and the differences in site characteristics. The proton budget calculations indicated a clear relationship between the net acidifying effect of nitrogen processes and the amount of N deposition. When the deposition increased also N processes became increasingly important as net sources of acidity. Sulphur budgets calculations indicated a net release of S from many ICP IM sites, indicating that the soils are releasing previously accumulated S. Similar results have been obtained in other recent European plot and catchment studies.

Typically 90% of incoming N deposition is retained by the soil. Nevertheless, moderate to high level of N deposition is a necessary factor for elevated nitrate concentration in runoff. A critical deposition threshold of about 8-10 kg N ha<sup>-1</sup> a<sup>-1</sup>, indicated by several previous assessments, was confirmed by the input-output calculations with the ICP IM data (Forsius *et al.* 2001), as well as using a larger European database including these sites (MacDonald *et al.* 2002). Soil organic horizon C/N ratio seems to give a reasonable estimate of the annual export flux of N for European forested sites receiving throughfall deposition of N up to about 30 kg N ha<sup>-1</sup> a<sup>-1</sup>. When stratifying data based on C/N ratios less than or equal to 25 and greater than 25, highly significant relationships were observed between N input and nitrate leached (MacDonald *et al.* 2002). It should also be recognized that N is usually the limiting nutrient in forest ecosystems, and thus sequestration of C (and the global C cycle) is closely linked to the N cycle and changes in deposition loads. Understanding the N cycle in forests is therefore essential also for determining the long-term source or sink for C in soils (Gundersen *et al.* 2006).



## Dynamic modelling

In a policy-oriented framework, dynamic models are needed to explore the temporal aspect of ecosystem protection and recovery. The critical load concept, used for defining the environmental protection levels, does not reveal the time scales of impacts/recovery. Dynamic models have been developed and used for the emission/deposition scenario assessment at selected ICP IM sites (e.g. Jenkins *et al.* 2003), and applied on large surface water databases compiled by ICP Waters and national programmes (Wright *et al.* 2005). These models are flexible and can be adjusted for the assessment of alternative scenarios of policy importance. These modelling studies have shown that the recovery of soil and water quality of the ecosystems is determined by both the amount and the time of implementation of emission reductions. According to the models, the timing of emission reductions determines the state of recovery over a short time scale (up to 30 years). The quicker the target level of reductions is achieved, the more rapidly the surface water and soil status recover. The model simulations also indicate that N emission controls are very important to enable the maximum recovery in response to S emission reductions. Increased nitrogen leaching has the potential to not only offset the recovery predicted in response to S emission reductions but further to promote substantial deterioration in pH status of freshwaters and other N pollution problems in some areas of Europe (Jenkins *et al.* 2003).

Wright *et al.* (2005) applied the MAGIC and SMART models to 12 acid sensitive surface water regions in Europe. The model results indicated that even after complete implementation of the Gothenburg Protocol and other current legislation, acidification with commensurate adverse biological effects will continue to be a significant problem in southern Norway, southern Sweden, the Tatras, the Italian Alps, and the Southern Pennines in the United Kingdom. More than 5% of the ecosystems in each of the regions evaluated would not meet the ANC criterion to protect sensitive aquatic organisms. Additional mitigation measures would be required in these regions to meet long-term European policy objectives. The model simulations also indicated that, as expected, the percent base saturation (%BS) of soils decreased during the long period of acidification of 1860–1980. Between 1980 and 2000, the large reductions in sulphur deposition appeared in most cases to be sufficient to stop the decrease in %BS but still insufficient to allow %BS to recover. The prognosis for the future indicated little or no recovery of base saturation in the soil, and in one of the modelled regions (the Tatra Mountains in Slovakia) the soil would continue to acidify.

Future global change introduces another uncertainty to the predictions of acidification recovery. In a joint modelling study of ICP IM and ICP Waters (conducted under the framework of the EU project EURO-LIMPACS) the relative sensitivity of different climate change related processes affecting acidification recovery was investigated (Wright *et al.* 2006). The results showed that several of the factors are of only minor importance (increase in partial pressure of CO<sub>2</sub> in soil air and runoff, for example), several are important at only a few sites (e.g. seasalts at near-coastal sites) and several are important at nearly all sites (e.g. increased DOC in soil solution and runoff). In addition changes in forest growth and decomposition of soil organic matter are important at forested sites and sites at risk of nitrogen saturation. The trials suggested that in future modelling of recovery from acidification should take into account possible concurrent climate changes and focus specially on the climate-induced changes in organic acids and nitrogen retention.

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### 3 Progress report on updated assessments of heavy metal loads and critical limits at ICP IM sites.

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#### 3.1

#### **Introduction**

Impacts of exceeded critical loads of heavy metal loads could exert deteriorating effects in terrestrial and freshwater systems exposed to leaching from upland soils. Metals have accumulated in soils and catchments over long time periods. At such sites, negative influences on the biological system would affect ecosystem functions. There are also human health aspects of heavy metals.

Investigations on heavy metals (HM) in ecosystems have taken place over the years with intentions to extend the understanding of relevant processes, to gather information for critical levels and critical loads and to provide data for modelling. At the ICP IM sites, investigations have focussed on pools and fluxes (Aastrup *et al.* 1995, Munthe *et al.* 1998, Ukonmaanaho *et al.* 2001). Input of pollutants is the basis for the effects related work under LRTAP Convention. In the LRTAP Convention Protocol 1998 on Heavy Metals, Cd, Hg and Pb were identified to be of priority concern, with Hg presently highlighted. The concentration levels, presently reached, are estimated to be high enough to cause damage at 5-25% of European sites (Rademacher 2001).

Relative changes for the period 1980-2000 are well presented by EMEP for the priority heavy metals (EMEP 2004). European lead and cadmium emissions and deposition have decreased substantially during the period. Also emissions of mercury have decreased but deposition of Hg to a lesser extent. The development of metal contents in forest soils is of interest in this situation.

The integrated monitoring approach, as carried out in the ICP IM programme, furnishes complete investigations on concentrations in several compartments and good possibilities for budget calculations. In the ongoing work, a number of controlled IM catchments are included. In this report we present total metal deposition to forests, input/output balance for catchments and metal trends in humus layers. Critical loads were calculated for some sites.

## Methods

The ICP IM programme furnishes complete determinations for estimating HM pools, fluxes and critical loads. This is achieved by the catchment approach with controlled input and output. Inputs are from direct atmospheric deposition (bulk deposition, BD), throughfall (TF) and litterfall (LF), although litterfall in some cases includes internally circulated elements. Output is provided from measurements of runoff at the catchment discharge station. Metal contents are followed in soils and biomass fractions.

Such determinations could be found for 13 ICP IM sites during years within the period 1997 – 2003. The IM sites included relate to 8 countries; Austria (1 site), Czech Republic (2 sites), Finland (2 sites), Germany (1 site), Latvia (2 sites), Lithuania (2 sites), Sweden (4 sites) and UK (1 site).

## Site description

Forest is dominating vegetation in most of the catchments but of various extents. One site, GB01, is covered by dwarf shrubs and grass. Soils vary from sorted sediments on sedimentary bedrock over morainic landscapes on igneous bedrock to sites with substantial catchment sub areas being peatland and lakes. Mineral soils dominate but in UK and Finland there are mainly peaty soils. Especially in Austria the dolomite and limestone furnish karstic conditions with complicated water flows.

## Results

### Metal deposition to forests

Metals stored in soils should be seen in relation to the history of much higher pollution loads in earlier decades. General ranges for recent EMEP annual deposition of Pb was estimated to 0.5-1 mg m<sup>-2</sup> in large parts of Europe with levels exceeding 5 mg m<sup>-2</sup> in some parts of Central Europe (EMEP 2004). The range for Cd in year 2000 was 0.01-0.05 mg m<sup>-2</sup> and for Hg 0.005-0.05 mg m<sup>-2</sup> with values for both metals reaching > 0.15 mg m<sup>-2</sup> in some regions. This corresponds to the bulk deposition measured at IM sites. Deposition has decreased, according to assessments by EMEP for the year 2000 deposition of Pb was reduced by 80-95 %, Cd 30-80% and Hg by 15-60 % in most countries compared to 1980.

However, deposition assessed by bulk deposition only provides low values and both throughfall (TF) and litterfall (LF) should be considered. Especially for mercury it can be assumed that uptake in vegetation from soil is negligible making TF + LF a clear deposition estimate. Although this may be less so for Pb and Cd, TF + LF has rough relevance as deposition estimates for these metals, too. This changes the amounts to higher levels. Flows by both LF and TF were substantially larger than bulk deposition to open fields, which clearly demonstrates the role of interception in canopies. EMEP estimations give values on lead deposition at 0.5-1 mg m<sup>-2</sup> while throughfall for Pb at IM sites was 0.6 – 3.0 mg m<sup>-2</sup> a<sup>-1</sup>. Estimated values on cadmium deposition were 0.01-0.05 mg m<sup>-2</sup> a<sup>-1</sup> to be compared with IM input by TF of 0.03-0.27 mg m<sup>-2</sup> a<sup>-1</sup>. Also litterfall has to be added to the input. For mercury a recent estimate for southern Swedish IM site SE04 was about 50 µg m<sup>-2</sup> a<sup>-1</sup> in TF + LF which is about seven times higher than the open field wet deposition at the site.

### 3.4.2

#### Metals in stream flow

Annual water discharges varied to a large extent between sites, which is of consequence for metal transports. High mountain sites subject to high amounts of precipitation and low evaporation had annual water discharges between 800 to 1400 mm irrespective of land cover of grassland or forest (AT01, DE01, GB01). East European lowland sites in conditions of rather low precipitation had stream water discharges of only 50-200 mm (CZ01, FI01, FI03, LV01, LV02, LT01, LT01, LT03). Swedish sites of the study and the Czech highland site CZ02 were intermediate with annual stream water flows between 280-520 mm.

The general range of stream water flows for Pb was found to be 0.04-0.3 mg m<sup>-2</sup> a<sup>-1</sup> with two exceptional sites, CZ02 and DE01, having much higher flows, i.e. 1.19 and 1.42 mg m<sup>-2</sup> a<sup>-1</sup>, respectively. Special circumstances, such as heavy storm flow and forest damage, provided these extreme values. The leaching could be compared to the depositions on 0.6 – 3.0 mg m<sup>-2</sup> a<sup>-1</sup>. For the time period of investigation when deposition loads had been much reduced, Pb levels were retained in the catchments to a degree of 16-97% of input by TF.

The range of Cd runoff at eight sites was 0.005-0.02 mg m<sup>-2</sup> a<sup>-1</sup> independently of large differences in water discharge. Corresponding Cd input was 0.03-0.26 mg m<sup>-2</sup> a<sup>-1</sup>. Cd is considered more mobile than Pb and outflow of Cd was between 4% and >100% of input by BD or TF. The high output values can be related to special conditions such as extremely high water discharge and forest damage. In more regular situations catchments clearly accumulate Cd.

The third metal considered was Hg with deposition values at 0.012-0.046 mg m<sup>-2</sup> a<sup>-1</sup>. Fluxes in the monitored streams were in the range of 0.001-0.007 mg m<sup>-2</sup> a<sup>-1</sup> and retention was 77-97% of total input. Hg deposition was at elevated levels making continuing accumulation in the system obvious. The dependence of Hg transport on organic matter is considered strong and thus movement of organic material is crucial. Although mercury is retained in catchments, the major environmental concern is the formation and transport of methyl-mercury in discharge areas. The role of forestry in this mobilisation has been highlighted lately.

### 3.4.3

#### Metal loads to forest humus layers

Although effective retention is prevalent on the catchment scale, the situation is different in the upper part of soils.

Table 3.1 Metal contents in southern Swedish mor layers (humus layers) in two different years as measured in IM sites.

	Lead µg g <sup>-1</sup>		Cadmium µg g <sup>-1</sup>		Mercury µg g <sup>-1</sup>	
SE04 Gårdsjön						
	1995	2003	1995	2003	1995	2003
Whole layer	102	87	0.35	0.35	0.36	0.41
F-layer	77	40	0.40	0.26	0.27	0.28
SE14 Aneboda						
	1996	2005	1996	2005	1996	2005
Whole layer	81	42	0.45	0.42	0.27	0.33
F-layer	74	23	0.39	0.32	0.23	0.25



Repeated measurements in recent decade of metal contents in the organic layers on top of the podsol soils of the southern Swedish IM sites showed a sharp decrease of lead contents (Table 3.1). The decrease was more pronounced in the F-layer, i.e. in the upper part of the humus layer, than in the organic layer as a whole. This can be expected as the upper layer is more readily affected by the input of new plant litter having lower metal contents. The subdivision of the humus layer is an easy way of improving sensitivity in order to get early detection of change, although the layer needs to have sufficient thickness for good sampling. For cadmium, a decreased level in the F-layer was detected but not in the humus layer as a whole.

Mercury showed no significant changes, neither in the whole layer nor in the F-layer. This means that reduction of atmospheric load is still insufficient to initiate an improvement of Hg levels in the topsoil.

In Sweden, Norway and the Czech Republic there are good national inventories of Hg in mor layers. Data from other countries are more scanty. In the Czech Republic the average level in humus layers was determined to  $0.68 \mu\text{g g}^{-1}$ , with local high values at  $>2 \mu\text{g g}^{-1}$  (Suchara and Sucharova 2000). Recent determinations in the IM site Neuglobsow (DE02) gave the value  $0.34 \mu\text{g g}^{-1}$  calculated on the basis of organic content, i.e. at the same level as values in southern Sweden. Hg has the least promising development of the three LRTAP Convention's metals with no signs of recovery even in top soils. Soil monitoring, however, only takes place in a few of the IM sites.

#### 3.4.4

#### Critical limit for mercury

The critical limit is the starting point in the critical load assessment. The risk assessment for Hg in the critical load manual deviates from those of Pb and Cd (UBA 2004). For the latter there is a generalized modelling approach valid for all soils in relation to the free ions, while for Hg there are empiric assessments for fish and separate assessments for total contents in humus layers. There is knowledge that atmospheric Hg is very effectively retained in humus layers. During the course of time a substantial storage has occurred reaching 3-10 times the pre-industrial level in various parts of Europe. A critical limit value has been estimated from indicative small-scale correlations of Hg to soil respiration in soil plots (Bringmark & Bringmark 2001). As lead and other factors are confusing in the relations for Hg in the field situation, experimental effect determinations have also been performed.

Experiments on southern Swedish humus layer materials gave an EC5 value at  $2 \mu\text{g Hg g}^{-1}$  (5% effect concentration). This was considered low enough to support indications from field studies of a critical limit at  $0.5 \mu\text{g g}^{-1}$  organic content bearing in mind the common practise to apply safety factors on measured experimental effect concentrations in risk assessments. Recent experimental determinations repeated for northern Swedish humus layer materials turned out more sensitive than the southern Swedish ones, yielding EC5 value at  $0.9 \mu\text{g g}^{-1}$ . It is hypothesized that carbon availability for micro organisms is a factor influencing sensitivity, as respiratory activity is higher in the north during the short non-winter time. The latter EC5 value is in the range of the highest Central European values found e.g. in the Czech Republic (Suchara and Sucharova 2000).

### 3.4.5

#### Critical load for Hg in humus layer

The calculation model for Hg described in the critical load manual (UBA 2004) is based on Meili *et al.* (2003). A prerequisite is the very strong association of Hg to organic matter with virtually no free HgII. The leaching of Hg is described by the relation between Hg contents in solid and dissolved organic matter.

$$[\text{Hg}]_{\text{diss}} = [\text{Hg}]_{\text{OM}} \cdot f \cdot \text{DOM}$$

$[\text{Hg}]_{\text{diss}}$	Hg in solution
$[\text{Hg}]_{\text{OM}}$	Hg in soil organic matter (OM)
DOM	dissolved organic matter
f	fractionation ratio between Hg in DOM and Hg in OM

Numerical example from the critical loads manual:

critical limit for Hg in soil organic matter,  $0.5 \mu\text{g Hg g}^{-1} \text{OM}$

DOM =  $70 \text{ mg l}^{-1}$

f = 1

$[\text{Hg}]_{\text{diss}} = 0.035 \mu\text{g l}^{-1}$ , for calculation of critical leaching by multiplying with water flux.

Critical leaching is equal to critical load under the assumption of steady state. The factor f has been assigned the value 1 as a result of research at IM sites (Åkerblom 2006). Water flux as well as DOM values have to be provided in mapping procedures. No data on metals are needed. At IM sites it is possible to do critical load calculations that can be put in relation to deposition assessments according to IM as well as EMEP. Existing metal mass balances can be observed for comparison. In most cases these will show unbalanced situations far from the theoretical future steady state assumed for critical load.

Total deposition of Hg estimated by TF+LF at IM sites clearly exceeded calculated critical loads for Hg at  $0.006\text{--}0.0016 \text{ mg m}^{-2} \text{ a}^{-1}$  (SE14, SE15, SE16). For Pb, the E-horizon was found to be just at risk level, while the humus layer was safe. Cd deposition was clearly below critical loads.

### 3.5

## Conclusions

In spite of decreasing loads, effective accumulation of Pb, Cd and Hg is still going on in small forest catchments. Actual accumulation is higher than deposition values given by EMEP due to dry deposition not fully included in the EMEP values. Throughfall and litterfall measurements provide alternative estimates for IM sites.

For the biologically important humus layers, there is now evidence of sharply decreasing Pb concentrations. There is also some evidence of decreasing Cd concentrations. In contrast, Hg shows no signs of decrease even in the humus layer. Experimental foundation for assigning possible risks from Hg on soil biota has been supported by new experiments. In combination with field assessments in soil plots we maintain the claim that humus layers are at risk at southern Swedish Hg levels. The situation should be closely monitored if possible in a larger number of IM sites, in order to detect improvement or deterioration.

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## 4 Target Load Functions: an update, including some climate change impacts

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### 4.1

#### Introduction

A modelling study of the Afon Hafren, UK (Hutchins & Jenkins 2006) reached some preliminary conclusions for the derivation of target load functions for acid deposition at ICP IM sites. For the Hafren, critical load estimates generated by a steady-state model (FAB) and by a dynamic model (MAGIC) are similar. Stream water responses to changes in deposition are rapid and under future deposition levels projected by Gothenburg Protocol will not be close to critical load exceedance (when considering a stream ANC target of 20  $\mu\text{eq l}^{-1}$ ). Nevertheless deposition target loads should be set as far into the future as deemed feasible as recovery may not be sustained in this particular catchment system. The catchment soils are poorly buffered and have always exceeded critical acidification thresholds. An initial analysis of the possible effects of climate change on recovery from acidification was undertaken using the method of Wright *et al.* (2006). Here, an extension of this work on target load functions in the context of climate change in mid-Wales, UK, is presented.

### 4.2

#### Effect of changing climate on MAGIC model predictions

Some projected effects of climate change on acidification recovery are demonstrated using MAGIC by means of (i) a one-by-one sensitivity analysis and (ii) an integrated approach whereby forecast sequences of some of the sensitive input parameters were altered simultaneously.

(i) The one-by-one sensitivity analysis followed the approach of Wright *et al.* (2006), in turn investigating the effects on model output of changes in: seasalt deposition, runoff, weathering rate, organic acids, partial pressure of carbon dioxide, forest growth, and soil organic matter decomposition.

(ii) The integrated approach included adjustments to streamflow, soil and stream temperature, organic acids, seasalt deposition inputs and soil decomposition. In the first two cases (streamflow and temperature) the effects were quantified using a combination of HadCM3 output and SDSM (statistical downscaling) (Wilby *et al.* 2001). Alterations to values for the latter three cases were guided by the approaches of Evans (2005) and Posch *et al.* (in press); investigations undertaken for the EU EURO-LIMPACS project. The adjustments and their impacts on model input and

parameterisation are summarised as follows. Stream flow remained unchanged. Soil temperature underwent a linear increase from 2000 to 2100 by a factor of 1.18. Likewise a linear increase (factor of 1.18) in seasalt deposition was applied from 2000 to 2100. Stream and soil DOC values ( $\text{mmol m}^{-3}$ ) were subject to a linear increase from 2000, being 50% higher in 2100. A linear increase in net decomposition rate ( $\text{mol C m}^{-2} \text{a}^{-1}$ ) from 0 in 2000 to 0.3 in 2100 was applied.

The results are displayed on Figure 4.1. It can be seen that the model is sensitive to future increases in soil decomposition (N release), having an adverse effect on the recovery of streamwater ANC (potentially violating critical levels once again before 2050) and soil base saturation. In the shorter term (up to 2050) increases in seasalt deposition appear likely to also have an adverse effect on ANC but this is unlikely to result in a decrease below  $20 \mu\text{eq l}^{-1}$ . Some more desirable effects are projected: increasing seasalt inputs are beneficial to soil base saturation and increasing weathering rates under higher temperatures will enhance recovery in both soils and stream water. When considering integrated climate change impacts on soil and stream quality, there is imperceptible impact on ANC although nitrate levels increase in the stream. Detrimental impacts on soil base saturation are seen and these impacts are ongoing and worsening through to 2100.

#### 4.3

### Effect of changing climate on target load functions

Critical load functions (CLF) and target load functions (TLF) are plotted on Figure 4.2. These were calculated as described by Hutchins & Jenkins (2006). Plots are analogous with the one published previously (Figure 4.4: Hutchins & Jenkins 2006) which includes the most stringent target load function, set for 2100. The likely effects of climate change are indicated. Sensitivity analysis illustrates the extent to which the target load (in deposition for year 2020) should be modified (Figure 4.2a). It is clear that the effects of soil decomposition serve to tighten the targets particularly in respect of nitrogen.

Other changes however, particularly those related to temperature and higher weathering rates, allow for more lenient targets. Consequently, when considering an integrated effect of climate change on model predictions, the target load function is likely not to change greatly (Figure 4.2b). The graph suggests that slightly higher S deposition would be acceptable.

#### 4.4

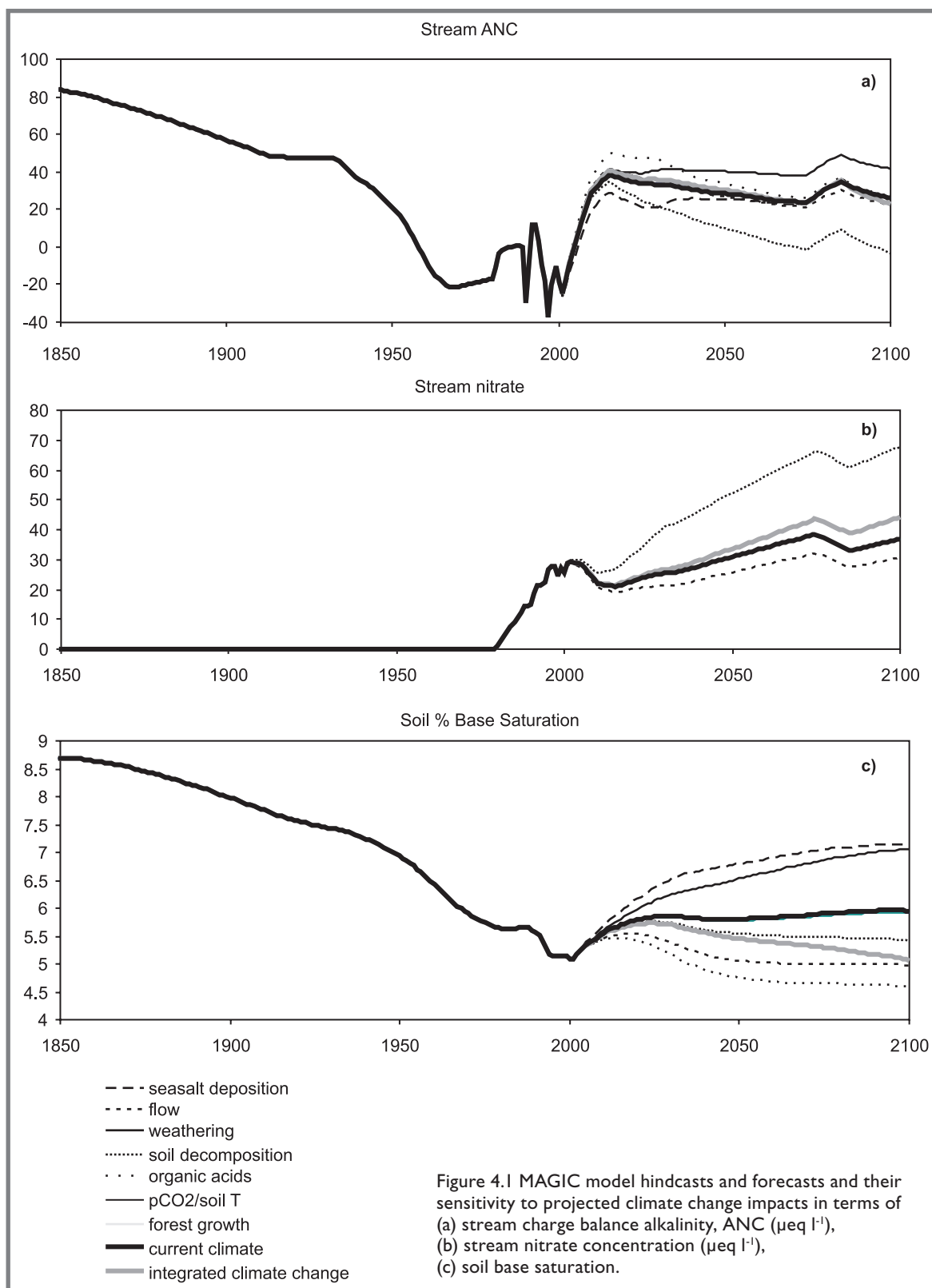
### Conclusions and future work

Research suggests that changes in climate will not have a large impact on mean annual stream water chemistry at the Afon Hafren, at least in terms of ANC and recovery from acidification. However, a holistic appraisal of future target loads should consider critical loads for eutrophication. Hutchins & Jenkins (2006) made a best estimate (catchment average) of the critical load for nutrient N of  $0.95 \text{ keq ha}^{-1} \text{a}^{-1}$  for Afon Hafren. Given the uncertainty regarding future catchment N dynamics and a projected elevation of stream nitrate concentrations attributable to climate change impacts (Figure 4.1b) imposing some further constraint on target N deposition would appear advisable at this stage.

It is acknowledged that episodic short term responses have an important impact on streamwater chemistry and the resilience of freshwater biota. This importance will increase in relative terms as systems recover from chronic acidification. The nature of such responses is likely to be very sensitive to climate change. There are likely to be changes in the severity of episodes and their predominant causal factors. Research



by Evans *et al.* (in press) has been undertaken to integrate these responses with the effects of long-term processes and changes in acid deposition at Afon Gwy, a site very near to Afon Hafren in mid-Wales, UK. For this purpose a two box version of MAGIC has been used. An appraisal of critical loads and target load functions for this site should be undertaken, together with an assessment of whether targets require modification under climate change projections. The analysis of target load functions should be carried out in a similar way to the one described above for the one-box Hafren MAGIC application.



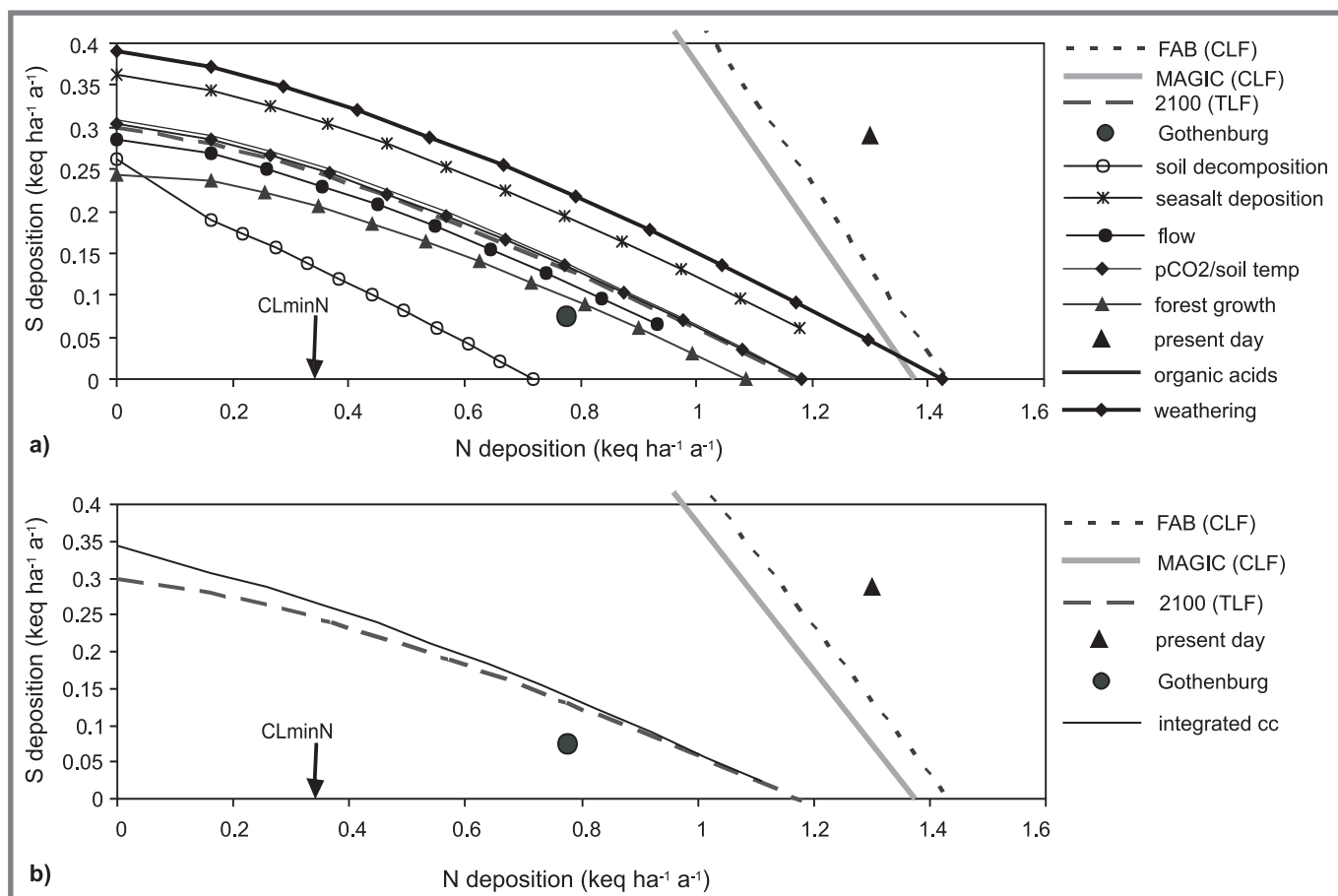


Figure 4.2 Critical load and target load functions, current and projected deposition under the Gothenburg protocol and (a) target load functions for 2100 under a one-by-one sensitivity analysis (after Wright *et al.* 2006) (b) target load functions for 2100 under an integrated assessment of climate change effects.

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## 5 Pine forest vegetation dynamics at ICP IM sites in Latvia

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### Abstract

In Latvia three integrated monitoring sites are established in pine stands in two different regions of the country. The aim of the current research was to study the changes in vegetation and environmental factors, and to characterize the regional differences in stand transformation processes of pine forests in Latvia during the period of 12 years. The data set of the present research includes species composition and abundance of tree, shrub, ground and bottom layers measured in three degree scale in 3 x 3 m squares (100 in each plot, three plots per site, three sites). Analysis of species frequencies and ordination of vegetation data was carried out. Ellenberg indicator values and direct measures of atmospheric deposits were used to explain vegetation dynamics.

Ordination revealed that there is a significant temporal gradient in vegetation dynamics from light nutrient poor and species poor forests to more nutrient rich, species more diverse and closed forests during 12 years period. This gradient is more pronounced in Rucava than in Taurene and shows that the process of vegetation transformation differs regionally. Relationships between vegetation dynamics and changes in Ellenberg indicator values and atmospheric deposition can be interpreted both as natural forest ageing and slight eutrophication but it is not possible to separate both influences based on the present data of 12 years period.

5.1

### Introduction

Dry pine forests are the most common and economically the most valuable forests in Latvia. At the same time pine stands of oligotrophic and oligo-mesotrophic sites have experienced substantial changes during the last 50 years. The total area of pine forests was 53.3 % of total area of forests in 1940 in Latvia, but it had decreased to 35.3 % in 2006 (area of oligotrophic forest site types *Cladinoso-Callunosa*, *Vacciniosa*, and *Myrtillosa* which are the most suitable for pine changed from 14.7 to 11.7 % during this period). The main reason for transformation of pine stands is anthropogenic influence, and amelioration of wet forests in particular. Presumably, also environmental dynamics has influenced the floristic composition of pine forests: increase of air temperatures (climate warming) and increase of atmospheric deposition (with maximum in the 60-80s of the last century). In Latvia, vegetation ruderalisation, graminification and fruticification are the processes that indicate the transformation of pine forest stands

under the influence of man (Laivinš 1998) and namely the slow gradual eutrophication of forests. Similar processes of forest environment eutrophication are referred from other boreonemoral regions, based on long term observations of species composition (Falkengren-Grerup 1986, 1989; Kuhn *et al.* 1987; Thimonier *et al.* 1992, 1995; Bobbink *et al.* 1998 etc.).

To characterize the transformation processes of pine forests (indicator species, deposition, critical loads etc.) intensive observations of vegetation in the Integrated Monitoring stations are of great value. Such investigations give insight into temporal dynamics of species frequencies and abundances, and reflect precisely the rate of eutrophication process.

In Latvia three integrated monitoring sites are established in pine stands in two different regions of the country. The aim of the current research was to study the changes in vegetation and environmental factors, and to characterize the regional differences in stand transformation processes of pine forests in Latvia during the period of 12 years.

## 5.2

# Material and methods

## 5.2.1

### Study area

There are two integrated monitoring stations in Latvia – Taurene (situated in the Vidzeme Upland) with one site (located in the basin of the forest stream falling into the Lake Taurene) and Rucava (situated in the Coastal Lowland) with two sites – Brušviti and Peši that are located in the basin of the Vārnupīte River. All three sites are established in pine stands representative for a region in 1993.

**Brušviti site** is located in 84 years old pine *Pinus sylvestris* stand with solitary birches *Betula pendula* classified as *Vaccinio vitis-idaea-Pinetum* var. *Calluna vulgaris* community. *Vaccinium vitis-idaea*, *Calluna vulgaris*, *Empetrum nigrum*, *Deschampsia flexuosa*, and *Melampyrum pratense* dominate in ground layer and *Pleurozium schreberi*, *Dicranum polysetum*, *D. scoparium*, *Cladina rangiferina* etc. dominate in moss layer.

**Peši site** is located in 50 years old pine *Pinus sylvestris* stand with solitary spruces *Picea abies* classified as *Vaccinio myrtilli-Pinetum* var. *Deschampsia flexuosa* community. *Vaccinium myrtillus*, *V.vitis-idaea*, *Deschampsia flexuosa*, *Pteridium aquilinum*, and *Melampyrum pratense* dominate in ground layer and *Hylocomium splendens*, *Pleurozium schreberi*, *Dicranum polysetum* etc. dominate in moss layer.

**Taurene site** is located in 56 year old pine *Pinus sylvestris* stand with solitary spruces *Picea abies* and birches *Betula pendula* classified as *Vaccinio myrtilli-Pinetum* var. *typicum* community. *Vaccinium myrtillus*, *V.vitis-idaea*, and *Melampyrum pratense* dominate in ground layer and *Hylocomium splendens*, *Pleurozium schreberi*, *Dicranum polysetum* dominate in moss layer.

Species abundances depend on the type and intensity of natural and anthropogenic disturbances. In Rucava as well as in Taurene pine stands are formed naturally. According to information received from local people there was a forest fire in Brušviti site in 1985. Consequences of the fire were present still in 1993 – stems of pines were still black and there were soil patches denuded of vegetation in places, and substantial increment of *Calluna vulgaris* was observed.

An improvement cutting of stand was carried out in Peši site in 1989-1990 (shortly before establishing of protection status in 1993). Therefore the stand has become thin in some places, and along with better light conditions abundance of light demanding forest fringe species (*Vicia cassubica*, *Lathyrus sylvestris*, *Pteridium*

*aquilinum*, *Calamagrostis arundinacea* etc.) has increased. Permanent plots were located in this site in order to avoid cutting passage corridors. No substantial natural or anthropogenic influences have been observed in Taurene site.

## 5.2.2

### Sampling design and vegetation records

Intensive vegetation plots were established based on the existing experience (Anon 1994, 1998; Bråkenhielm 1992; Kleemola, Söderman 1993). Three plots of 30 to 30 meters size (A, B and C) were established in each of the monitoring sites. The edges of plots were oriented in north-south and east-west directions. In Brušviti the stand was homogeneous and plots were oriented so that their corners were connected (Figure 5.1). In Taurene and Peši pine stands were heterogeneous (in Taurene this was caused by articulated relief; in Peši there were large gaps in tree layer, the net of paths and other anthropogenic influences) and plots were placed in a distance from one another. The corners and plot centre were marked by 1 meter tall and 6 x 5 cm thick wooden sticks.

Permanent plot (marked as VMP in Figure 5.1) edges were divided into 3 meters long segments marked by wooden sticks that were numbered in the south-north direction with numbers (from 1 to 10) and in the east-west direction with letters (A, B, C...J). Vegetation was recorded in each 3 x 3 m square (100 in total, marked as IVP in Figure 5.1) that were marked in the field by nylon bands stretched among numbered wooden sticks.

In each permanent plot (30 to 30 meters) one circular plot with 15 meters radius was established for tree layer records (marked as FSSP in Figure 5.1). Shrub layer was recorded in 10 circular plots with 1 m in radius that were located randomly (SSP in

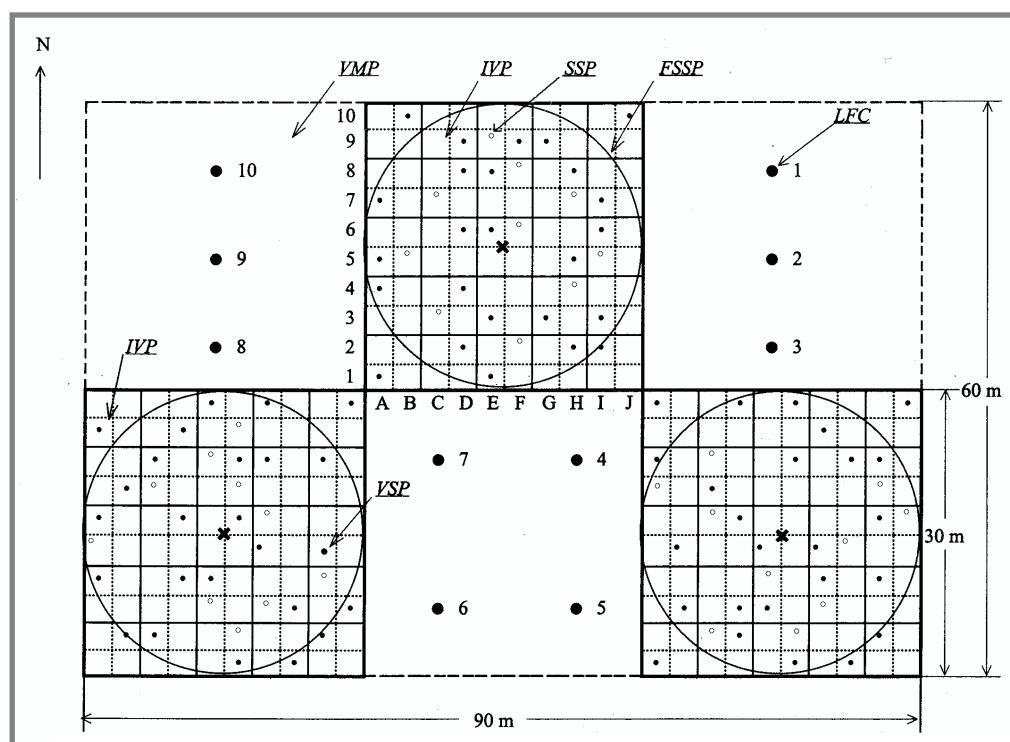


Figure 5.1 Intensive vegetation monitoring sampling design. VMP- vegetation monitoring plot (30 x 30 m); FSSP – tree layer record plot; IVP – flora mapping square (3 x 3 m); VSP – field and bottom layer record square (0.5 x 0.5 m); SSP – shrub layer record plot (R = 1 m); LFC – litterfall box.

Figure 5.1). Field and bottom layer records were made in 25 0.5 x 0.5 m squares (VSP in Figure 5.1) that were placed randomly.

Vegetation records in 3 x 3 m squares and estimates of vegetation cover in shrub, field and bottom layer were carried out once in three years, but taxation of tree layer was done once in six years.

The present work deals with analysis of 12 year dynamics of species composition and abundances in 3 x 3 m squares (IVP in Figure 5.1) (records were done in 1994, 1997, 2000, 2003, and 2006).

In 3 x 3 m squares all species (trees, shrubs, herbs, mosses, lichens) were recorded and cover of each species was estimated visually in three degree scale: 1 – cover of species is less than 1 %, 2 – cover of species is 1 – 25 %, 3 – cover of species exceeded 25 %.

These data were used to calculate species frequency in permanent plots A, B and C (there were 100 3 x 3 m squares in each permanent plot) and for the site in total (n = 300).

### 5.2.3

#### Environmental variables

Direct measures of environmental variables and species indicator values (Ellenberg *et al.* 1992) were used to explain variation and dynamics of vegetation in the course of time. Ellenberg indicator values were calculated in two ways. To calculate values for each square (3 x 3 m) only species presence/absence was used. For the calculation of mean value for a plot (30 x 30 m) weighted values were calculated using a sum of species cover (cover was estimated in the field in 3 degree scale).

Direct measurements were precipitation amount (mm) and concentration of ions in precipitation ( $\text{SO}_4^{2-}$ ,  $\text{NO}_3^-$ ,  $\text{NH}_4^+$ ,  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^+$ ,  $\text{H}^+$ ). The measurements were done in Rucava and in Taurene starting 1995 both in open field (bulk) and under tree canopy (throughfall). For details see integrated monitoring manual published on the internet <http://www.lva.gov.lv/monitor/>.

### 5.2.4

#### Data analysis

$Z_\alpha$  criterion was used ( $z_{0.05}$ ) to evaluate differences in species frequency in the period from 1994 to 2006 (Arhipova, Bālina 2003). Indicator species analysis was performed to reveal differences in species composition among all monitoring sites (Dufrene, Legendre 1997). Floristic diversity was analysed using the species richness and Shannon-Wiener diversity index. To use these parameters in species poor vegetation types, accuracy in species records and determination is of particular importance. Lack of precision was possible for lichen and moss layer in current data set. For that reason initially we calculated species richness and diversity index both for the full data set and for reduced data set. In the reduced data set all *Cladonia* and *Cladina* species were merged as *Cladonia sp.* and *Cladina sp.*, and several species that were hard to determine in the field or their occurrence was highly dependent on seasonal phenomena or biological peculiarities (*Monotropa hypopitys*, *Neottia nidus-avis* etc.) were eliminated. Results from both data sets indicated the same dynamics of floristic diversity in the period from 1994 to 2006 (Figure 5.2). As the common species are the best indicators of environmental changes, the reduced data set was used in the present research in all analyses.

To evaluate statistical significance for differences in floristic diversity among sites Mann-Whitney U test was performed (non-parametric test was used because variances of the data sets were not homogeneous according to Levene's test).



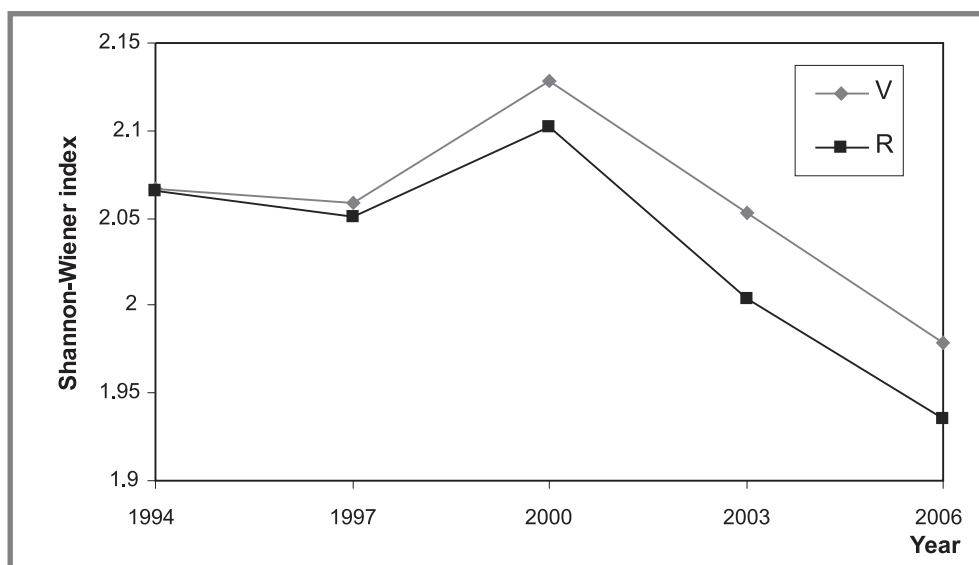


Figure 5.2 Shannon-Wiener diversity index values in different years in Taurene. V – complete data set (all species); R – reduced (without rare and taxonomically difficult species) data set.

To reveal inner diversity among all three monitoring sites detrended correspondence analysis (Hill, Gausch 1980) was performed using all 3 × 3 m squares (in total 4500) in one matrix (no down weighting of rare species was applied). Ordination axes were correlated with Ellenberg indicator values.

To reveal if there are directed changes in vegetation in each site, we followed the approach used by Nygaard and Ødegaard (1999). Ordination was performed for each monitoring site separately using all 3 × 3 m squares including all observation years in one data set. Down weighting of rare species was applied in all sites with exception of Taurene site where species diversity was the least and down weighting caused too strong influence on ordination of one species (*Pinus sylvestris*). DCA ordination plot scores for each year were analysed and significance of changes was evaluated by the non-parametric Wilcoxon signed rank test.

Software packages SPSS for Windows and Pcord 4 (McCune, Mefford 1999) were used for the analyses.

**Nomenclature** for vascular plants: Gavrilova, Šulcs, 1999; for mosses: Ābolina, 2001; for lichens: Piterāns, 2001.

### 5.3

## Results

#### 5.3.1

### Comparison of three monitoring sites

In total 96 species (vascular plants, mosses and lichens) were recorded in three monitoring sites in 5 observation years. Floristic similarity of Taurene site was 0.63 (Sørensen similarity index) both with Peši site and Brušviti site, but both last sites were less similar to each other (Sørensen similarity index is 0.49). From species with high constancy only *Goodyera repens* was unique for Taurene. All other species recorded only in Taurene site were sporadic (for example *Juniperus communis*, *Poa pratensis*). Peši and Brušviti sites had more unique species if compared to Taurene. Constant species that were not present in Taurene but constant in Brušviti or Peši were *Deschampsia flexuosa*, *Calamagrostis arundinacea*, *Scleropodium purum*, and *Empetrum nigrum*.

Indicator species analysis (after Dufrene, Legendre 1997) revealed several species that have significantly higher abundance in one site in comparison with two other sites (Table 5.1).

Table 5.1 Indicator species analysis (after Dufrene, Legendre 1997) for three monitoring sites in 1994, T – Taurene, P – Peši, B - Brušviti.

Species	Constancy, %			Mean cover (in degrees)			Indicator value	P
	T*	P	B	T	P	B		
Taurene								
Hylocomium splendens	100	73	63	2.6	1.9	1.8	51.3	0.001
Picea abies	54	9	12	2.2	1.7	1.7	41.6	0.001
Dicranum polysetum	96	80	86	1.4	1.4	1.1	38.2	0.001
Vaccinium myrtillus	98	87	70	2	2.5	2.4	34.6	0.001
Festuca ovina	13	4	-	1	1.2	-	10	0.001
Peši								
Deschampsia flexuosa	-	100	41	-	2.8	1.2	85.1	0.001
Melampyrum pratense	76	99	83	1.4	2.0	1.3	47.5	0.001
Pleurozium schreberi	100	100	99	2.7	2.9	2.8	34.6	0.001
Trientalis europaea	2	34	-	1	1.4	-	32.6	0.001
Calluna vulgaris	-	26	-	-	1.2	-	26	0.001
Maianthemum bifolium	2	23	-	1	1.2	-	21.2	0.001
Quercus robur	5	25	2	1	1	1	19.8	0.001
Pteridium aquilinum	5	24	-	1.7	1.6	-	19.2	0.001
Ptilia crista-castrensis	28	28	-	1	1.4	-	16.5	0.001
Betula pendula	20	32	13	1	1.2	1.6	15.5	0.001
Luzula pilosa	29	29	-	1	1.1	-	15.2	0.001
Frangula alnus	2	13	-	1	1.1	-	11.1	0.001
Brušviti								
Calluna vulgaris	35	15	97	1.2	1.3	2.1	74.8	0.001
Empetrum nigrum	-	-	30	-	-	1.3	30.3	0.001
Dicranum scoparium	35	12	48	1	1	1.2	27.5	0.001
Cladina arbuscula	-	-	4	-	-	1	4	0.001
Vaccinium uliginosum	2	-	4	1.4	-	1.7	3	0.005

DCA ordination diagram and Ellenberg indicator values correlation with ordination axes showed that each site was different from other sites (Figure 5.3). Peši site was the richest in nutrients and with better moisture conditions. Also species diversity was slightly higher and Ellenberg indicator values for temperature showed warmer conditions than for other sites.

Taurene site was associated with higher continentality figures reflecting differences in species composition – several oceanic species (*Deschampsia flexuosa*, *Scleropodium purum*) were not present in Taurene site. Brušviti site differed from others by better light conditions.

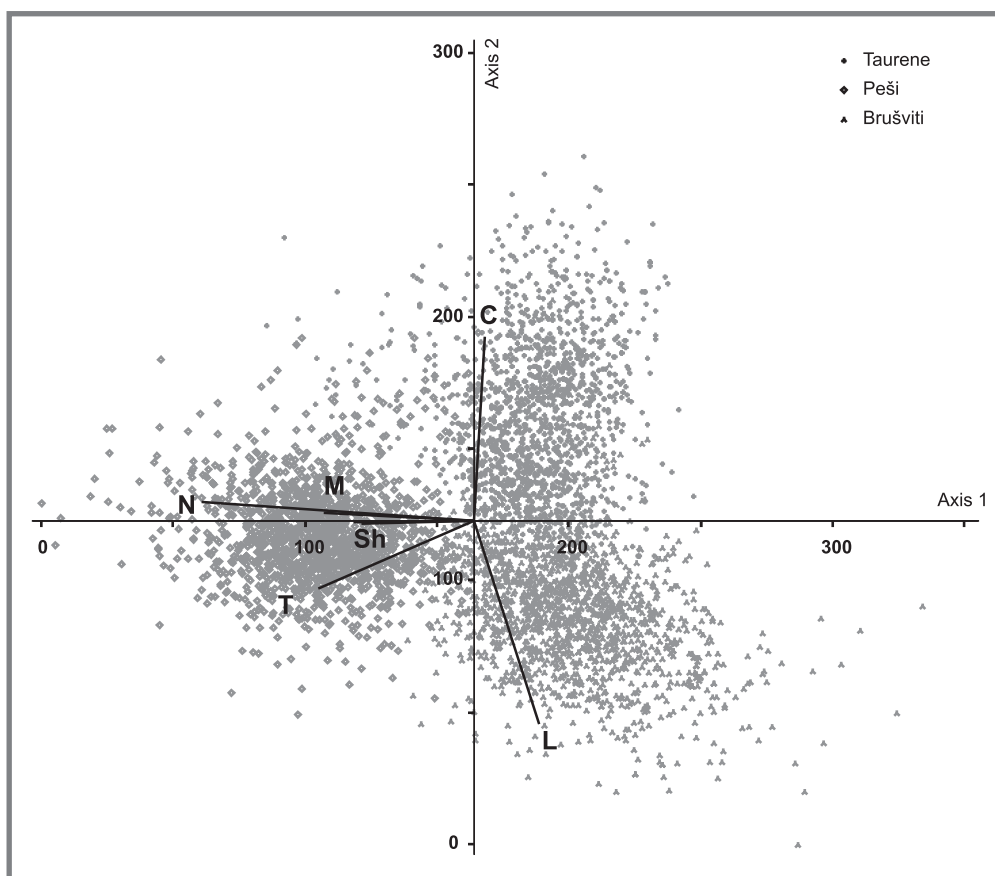


Figure 5.3 DCA ordination of the complete data set ( $n = 4500$ ). Sh – Shannon-Wiener diversity index; Ellenberg's indicator values: M – moisture, N – nitrogen, T – temperature, C – continentality, L – light. ( $\lambda_1 = 0.25$ , length of gradient 3.47;  $\lambda_2 = 0.17$ , length of gradient 2.57;  $\lambda_3 = 0.11$ , length of gradient 1.99; total variance (inertia) is 5.24, determination coefficient between ordination space and original multidimensional space is 0.54).

Table 5.2 Correlations of DCA ordination axis with Ellenberg indicator values and species diversity figures ( $n = 4500$ , \*\*  $p = 0.01$ ).

Value	Axis 1	Axis 2	Axis 3
Number of species	-0.51**	-0.06**	0.12**
Shannon index	-0.49**	-0.07**	0.10**
Light	0.37**	-0.66**	0.23**
Temperature	-0.58**	-0.38**	0.16**
Continentality	0.15**	0.63**	-0.21**
Moisture	-0.57**	0.14**	0.11**
Reaction	-0.32**	0.32**	-0.09**
Nitrogen	-0.77**	0.20**	-0.22**

### 5.3.2

#### Changes in species diversity

Shannon-Wiener index and species numbers were different in all three sites. Mann-Whitney U test showed that the number of species and Shannon-Wiener index was significantly different among all sites both with data from all observation years (Table 5.3) and for each year separately (not shown). The highest values of both diversity measures were in 2000 for Taurene and Brušviti, and in 2003 for Peši (Figure 5.4). Average differences among years were 1.3 for species number and 0.2 for Shannon-Wiener index in all sites.

Table 5.3 Comparison of floristic diversity among three monitoring sites (Mann-Whitney U test results)

Site	Mean number of species per 3x3 m square (n = 1500)	Shannon-Wiener index for 3x3 m square		
		Mean value for 3x3 m square	z-value (n = 1500)	Significance
Taurene (all years) <sup>a</sup>	8.6	2.03	-28.99 <sup>b</sup> -9.36 <sup>c</sup>	0.001 <sup>b c</sup>
Peši (all years) <sup>b</sup>	10.7	2.25	-28.99 <sup>a</sup> -21.29 <sup>c</sup>	0.001 <sup>a c</sup>
Brušviti (all years) <sup>c</sup>	9.01	2.09	-9.36 <sup>a</sup> -21.29 <sup>c</sup>	0.001 <sup>a b</sup>

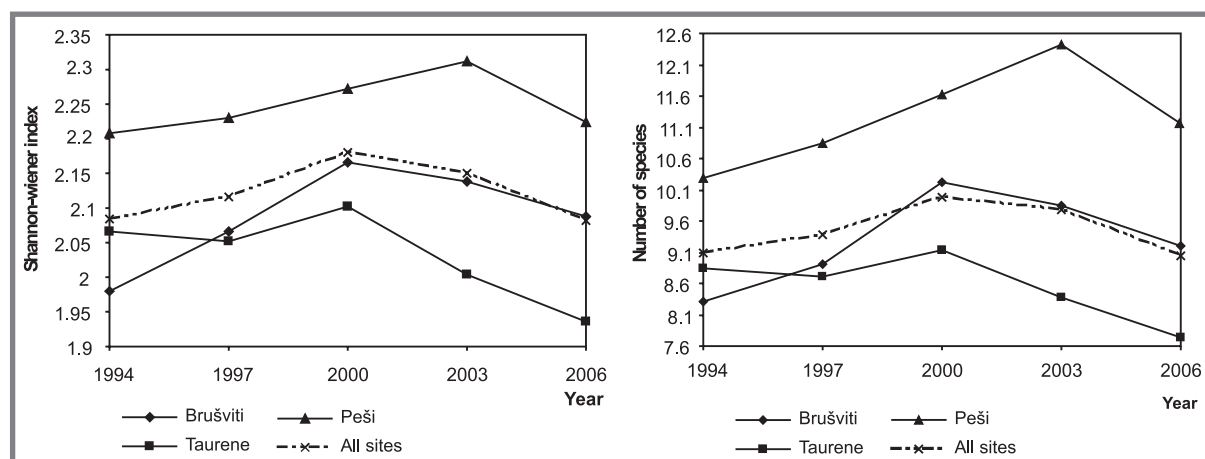


Figure 5.4 Dynamics of Shannon-Wiener index and mean species numbers in three monitoring sites in the 12 year period.

### 5.3.3

#### Changes in species frequency

Dynamics of species frequency between 1994 and 2006 was analysed. Statistical significance of changes was determined using  $z_a$  test (Arhipova, Bālina, 2003), and values are shown in Appendix 1. A 12 year period is a very short time in a medium aged pine forest (41-80 years old). Nevertheless, during the period several vegetation transformation tendencies were observed.

Proportion of *Picea abies* has increased in all plots both in tree and shrub as well as in field layer (Figure 5.5). The increment is more expressed in Peši and Taurene site than in Brušviti site (Appendix 1). With increasing role of spruce in forest stand light conditions become worse and abundance of light demanding species such as *Populus tremula* and *Betula pendula* decreases in shrub layer. They are replaced by *Frangula alnus* and *Sorbus aucuparia* that are characteristic for spruce forest shrub layer (Appendix 1).

Field and bottom layer enriches in *Vaccinium myrtillus*, *Maianthemum bifolium*, *Trientalis europaea*, *Hylocomium splendens*, *Brachythecium oedipodium* mostly on the expense of *Vaccinium vitis-idaea* and partially also *Melampyrum pratense*.

Species composition and abundance in the field layer experienced rapid changes after the forest fire in Brušviti. Abundance of *Cladonia* species (*Cladonia arbuscula*, *C. stellaris*) growing on soil and *Calluna vulgaris* decreased and *Deschampsia flexuosa* and *Vaccinium myrtillus* expanded in the field layer and *Hylocomium splendens*, and *Dicranum polysetum* in the bottom layer.

To conclude, oligotrophic and light demanding pine forest species have decreased in all monitoring sites during a 12 year period. They are *Calluna vulgaris*, *Festuca ovina*, *Vaccinium vitis-idaea*, *Dicranum scoparium* etc. Frequency of several mesotrophic species has increased, for example, *Picea abies*, *Vaccinium myrtillus*, *Maianthemum bifolium*, *Cirriphillum piliferum*, *Brachythecium oedipodium*.

The most conspicuous was invasion of oceanic temperate moss species *Scleropodium purum* in the monitoring sites of the Coastal Lowland (Brušviti, Peši). The number of occurrences of this species (total number of 3 x 3 m squares where it was recorded) has increased 12-fold in Brušviti and 4-fold in Peši in a period of 12 years.

#### 5.3.4

### Vegetation ordination

DCA ordination of each site separately (using all 3 x 3 m squares from all observation years) showed that overall vegetation heterogeneity was quite low – length of gradient was less than 2 standard deviations in ordinations of Peši and Taurene data sets and 2.8 standard deviations in ordination of Brušviti site. Ordination revealed both the inner heterogeneity among plots (in each site there were three plots: A, B, and C) and temporal vegetation dynamics. Ordination of Brušviti site is shown in Figure 5.6 for illustration.

In Brušviti site the plot B was quite different from plots A and C – Ellenberg indicator values showed that it is less nutrient rich with less species diversity and with better light conditions. DCA ordination of Peši site showed that inner heterogeneity of 30 x 30 m squares was not so pronounced as in Brušviti site. Plots B and C were very similar with each other and plot A differed a little bit in nutrient availability. DCA ordination with Taurene site revealed the same pattern as in Peši site. Two plots were quite similar (A and B) but the C plot differed from others. It supported less dwarf shrubs (*Vaccinium vitis-idaea*, *V. myrtillus*, *Calluna vulgaris*) but more herbs which are less light demanding but with higher requirements to soil nutrient availability (*Luzula pilosa*, *Pteridium aquilinum*). Light conditions were the best in plot B, but nutrient availability was best in plot C.

For all ordinations the first axis can be interpreted as nutrient gradient – correlations for the first axis was the highest with Ellenberg indicator value of nitrogen (Table 5.4).

Temporal vegetation dynamics (expressed as correlations between years of observation and ordination axes) was the most pronounced in Brušviti site, less in Peši site and very weak in Taurene site. Nevertheless, there were significant displacement of squares along the first and second axis in all sites and in Peši site also along the third axis in the course of time (Figure 5.7 & 5.8, Table 5.5).

The displacement along axis was associated with increased nitrogen values and to a lesser extent also with moisture, reaction and species diversity values and decreased light values along these axis in all three sites.

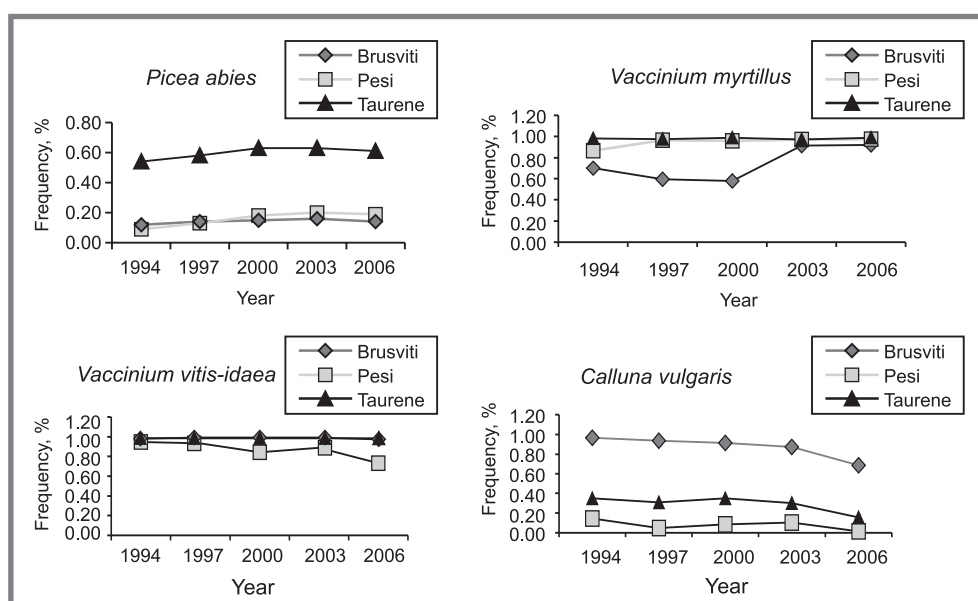


Figure 5.5 Dynamics of species frequency in the integrated monitoring plots.

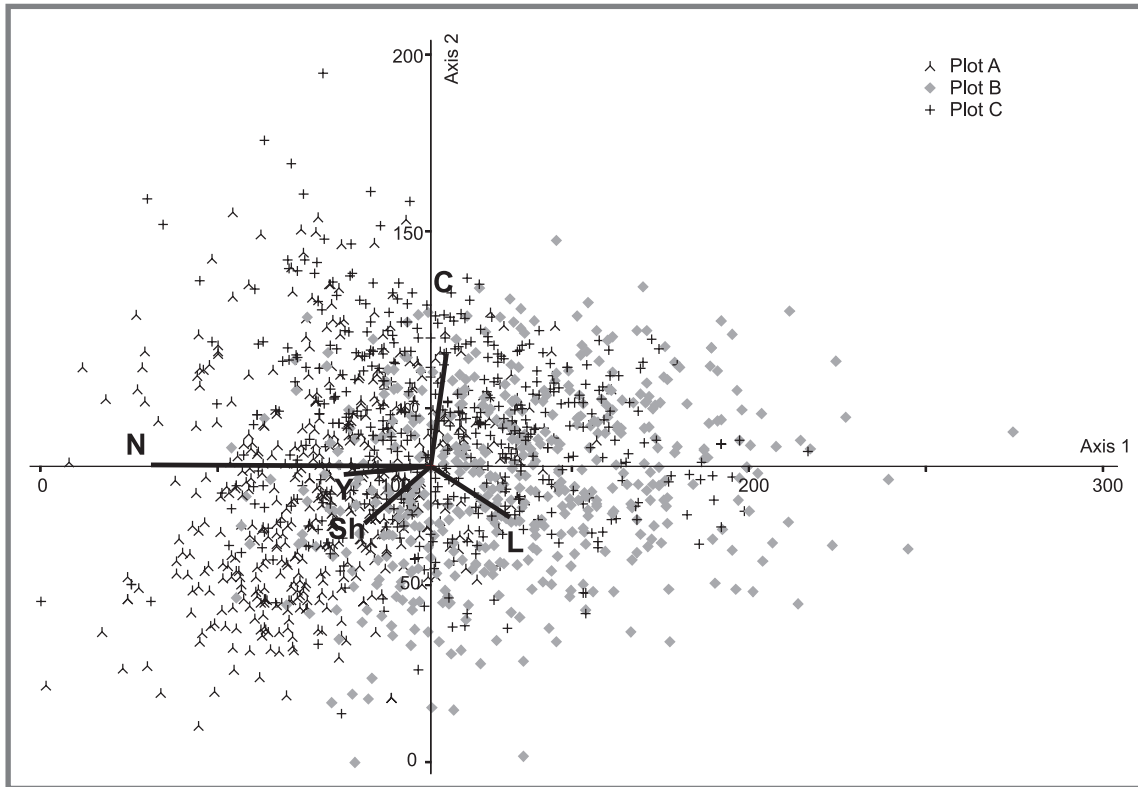


Figure 5.6 DCA ordination of Brušviti site (individual 3 x 3 m squares in all observation years are used). Sh – Shannon-Wiener index, Y – observation year, Ellenberg indicator values: L – light, C – continentiality, N – nitrogen.

Table 5.4 DCA ordination parameters and Spearman rank correlations among axes and site variables for three ordinations using 3 x 3 squares (for each site n = 1500; \* p = 0.01).

Variables	Brušviti			Peši			Taurene		
	Axis 1	Axis 2	Axis 3	Axis 1	Axis 2	Axis 3	Axis 1	Axis 2	Axis 3
Eigenvalue	0.13	0.09	0.07	0.09	0.08	0.07	0.17	0.12	0.11
Length of gradient	2.75	1.36	1.43	1.20	1.71	1.31	1.28	1.67	1.17
Total inertia	0.85			1.00			3.10		
r <sup>2</sup> between original and ordination space	0.60			0.39			0.43		
Correlations with axis									
Y – year of observation	-0.40*	-0.14*	-0.22*	0.27*	0.07*	0.27*	0.09*	0.19*	-0.10*
Sh – Shannon index	-0.31*	-0.37*	0.19*	0.24*	0.10*	-0.17*	0.05	-0.21*	0.04
Sp – number of species	-0.31*	-0.14*	0.18*	0.11*	0.17*	-0.21*	0.28*	-0.08*	-0.01
L – light	0.35*	-0.23*	0.07*	-0.21*	-0.08*	0.32*	-0.62*	-0.40*	0.05
T – temperature	-0.01	-0.14*	0.12*	-0.04	0.43*	-0.25*	-0.35*	-0.33*	-0.04
C – continentality	0.21*	0.43*	-0.25*	-0.21*	0.06*	0.16*	0.28*	0.36*	-0.22*
M – moisture	-0.19*	-0.26*	0.28*	0.26*	-0.01	-0.21*	0.43*	0.08*	-0.26*
R – reaction	-0.29*	-0.25*	-0.22*	0.19*	0.12*	0.14*	0.48*	0.11*	-0.04
N – nitrogen	-0.69*	0.01	0.23*	0.38*	0.24*	-0.41*	0.50*	0.12*	0.02



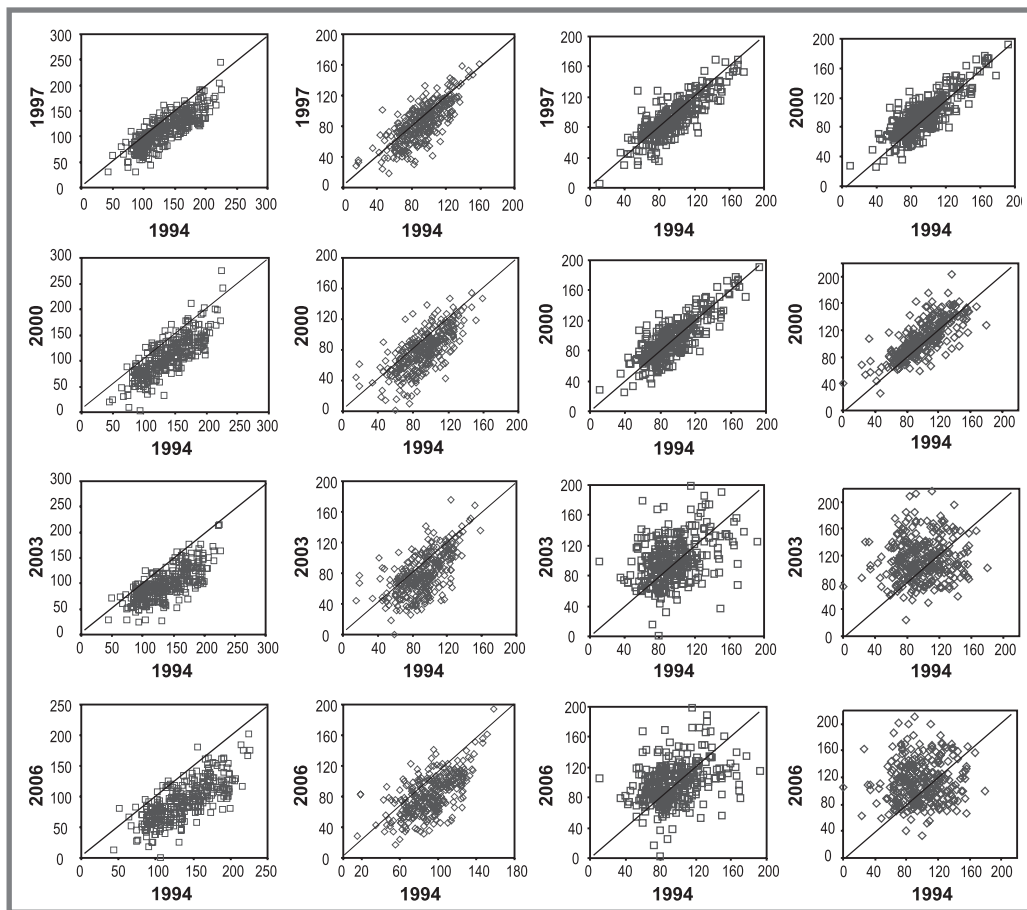


Figure 5.7 Changes in position of squares along the DCA axis I (left) and axis 2 (right) among the first and subsequent observation years in Brušviti site (two first columns on the left) and Taurene site.

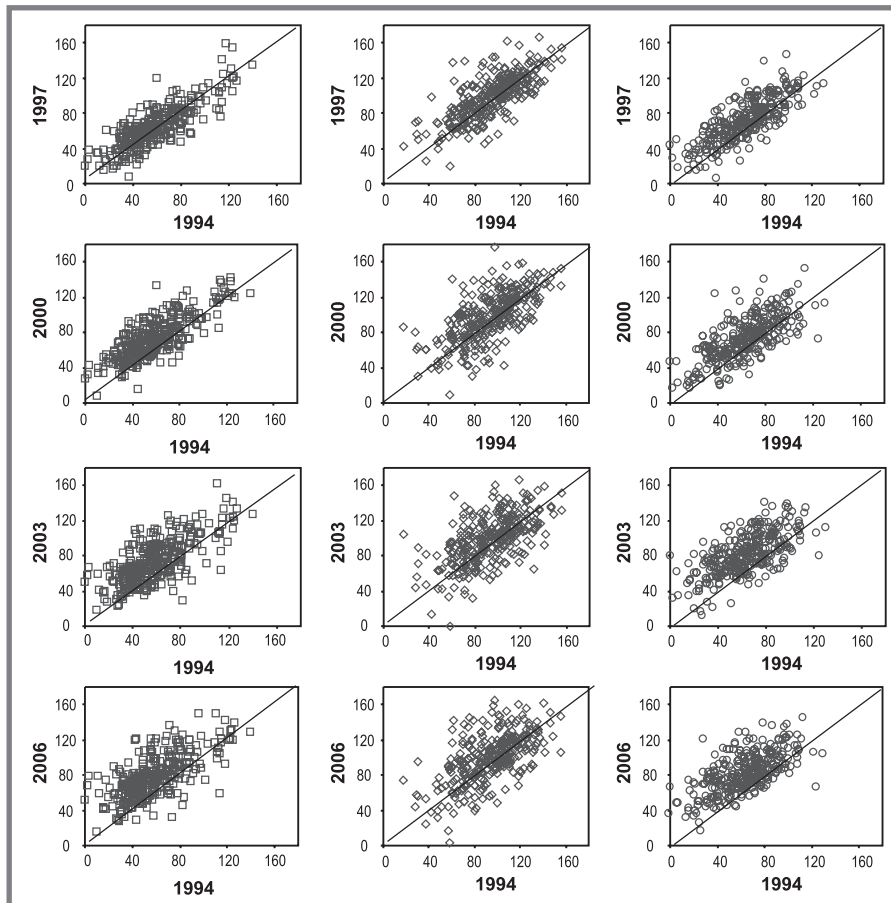


Figure 5.8 Changes in position of squares along the DCA axis I (left), axis 2 (middle), and axis 3 (right) among the first and subsequent observation years in Peši site.

Table 5.5 Median change of scores of squares in DCA ordination for three monitoring sites in different years (Wilcoxon signed rank test, n = 100 for each site).

DCA axis	Years	Brušviti			Peši			Taurene		
		Median change (in S.D. units)	Test statistics (Z)	P value	Median change (in S.D. units)	Test statistics (Z)	P value	Median change (in S.D. units)	Test statistics (Z)	P value
1	1994-1997	-0.23	-13.37	0.001	0.06	-6.40	0.001	-0.04	-4.22	0.001
1	1994-2000	-0.31	-13.98	0.001	0.15	-12.25	0.001	0.00	-0.42	0.670
1	1994-2003	-0.40	-14.83	0.001	0.15	-10.80	0.001	0.04	-2.62	0.009
1	1994-2006	-0.46	-14.87	0.001	0.19	-12.67	0.001	0.03	-2.16	0.031
2	1994-1997	-0.07	-6.23	0.001	0.05	-4.24	0.001	0.02	-2.73	0.006
2	1994-2000	-0.12	-9.71	0.001	0.05	-3.67	0.001	0.07	-6.65	0.001
2	1994-2003	-0.12	-8.73	0.001	0.06	-3.95	0.001	0.16	-6.31	0.001
2	1994-2006	-0.09	-6.97	0.001	0.06	-3.70	0.001	0.18	-6.07	0.001
3	1994-1997	0.00	-0.07	0.94	0.07	-7.48	0.001	0.00	-0.34	0.740
3	1994-2000	0.06	-5.79	0.001	0.09	-8.19	0.001	0.00	-1.14	0.260
3	1994-2003	0.13	-9.25	0.001	0.19	-11.92	0.001	-0.02	-3.23	0.001
3	1994-2006	0.12	-8.53	0.001	0.18	-12.76	0.001	-0.01	-2.05	0.041

### 5.3.5

#### Atmospheric deposition

Amount of deposits and their trends during the period of 11 years (1995-2005) were analysed for vegetation period (April-September), winter period (October-March) and for total year. Amount of S, Ca, Mg, K ions was higher in throughfall water than in bulk precipitation both in Rucava and Taurene. Such differences in ion concentration between open place and under forest canopy have been observed also in other regions (Simon, Westendorf 1991; Frey, Palo 1993; Aamlid 1994 etc.). It means that high amount of pollutants is concentrated in the air in the form of aerosols that settle on leaves and branches of trees and are washed off by rain. The only exception was nitrogen ions both in the form of nitrates and ammonium – higher amounts were recorded in bulk precipitation than in throughfall water (Figure 5.9).

Table 5.6 Trends for precipitation amount, precipitation acidity and amount of pollutants in the period (statistically significant figures in bold).

Variable	Rucava				Taurene			
	Bulk		Throughfall		Bulk		Throughfall	
	t	p	t	p	t	p	t	p
Perticipation_summer	-0.23	0.408	-0.70	0.242	1.25	0.106	<b>2.10</b>	0.018
Perticipation_winter	1.17	0.122	1.01	0.156	<b>1.96</b>	0.026	0.55	0.293
Perticipation_year	1.01	0.156	1.32	0.093	<b>1.79</b>	0.037	<b>1.79</b>	0.037
PH_summer	0.70	0.242	1.56	0.059	1.01	0.156	<b>2.10</b>	0.018
PH_winter	<b>1.87</b>	0.031	1.48	0.069	<b>2.57</b>	0.005	<b>2.81</b>	0.003
PH_year	<b>1.79</b>	0.037	1.32	0.093	<b>2.56</b>	0.005	<b>2.74</b>	0.003
SO4-S_summer	-1.00	0.156	<b>-2.10</b>	0.018	-1.50	0.069	<b>-2.00</b>	0.051
SO4-S_winter	-0.70	0.242	1.17	0.122	-1.32	0.093	-1.48	0.069
SO4-S_year	-1.48	0.069	-0.23	0.408	<b>-1.79</b>	0.037	-1.09	0.137
NO3-N_summer	<b>-1.96</b>	0.026	<b>-2.10</b>	0.018	<b>-2.88</b>	0.002	<b>-3.19</b>	0.001
NO3-N_winter	-0.23	0.406	1.01	0.156	0.23	0.408	-1.47	0.069

NO3-N_year	<b>-1.79</b>	0.037	0.38	0.349	<b>-2.10</b>	0.018	<b>-2.26</b>	0.012
NH4-N_summer	<b>-2.10</b>	0.018	<b>-3.04</b>	0.001	-0.70	0.242	-0.70	0.242
NH4-N_winter	<b>-1.79</b>	0.037	<b>-2.41</b>	0.008	-0.54	0.293	<b>-1.79</b>	0.037
NH4-N_year	<b>-2.41</b>	0.008	<b>-2.28</b>	0.002	-0.08	0.469	-0.86	0.196
Ca_summer	0.38	0.349	0.86	0.196	0.08	0.469	1.48	0.069
Ca_winter	0.23	0.408	<b>1.79</b>	0.037	-1.01	0.155	0.39	0.349
Ca_year	-0.54	0.293	<b>1.79</b>	0.037	-0.08	0.469	0.86	0.196
Mg_summer	1.16	0.122	1.52	0.064	0.26	0.394	<b>3.31</b>	0.001
Mg_winter	0.26	0.392	<b>1.69</b>	0.045	-0.27	0.393	0.26	0.394
Mg_year	0.09	0.464	<b>1.70</b>	0.045	-0.44	0.327	<b>2.16</b>	0.016
K_summer	-1.32	0.093	0.54	0.293	-1.48	0.069	<b>3.50</b>	0.001
K_winter	-0.08	0.469	<b>2.10</b>	0.018	1.01	0.156	1.63	0.051
K_year	-1.32	0.093	0.54	0.293	-1.56	0.059	<b>2.72</b>	0.003

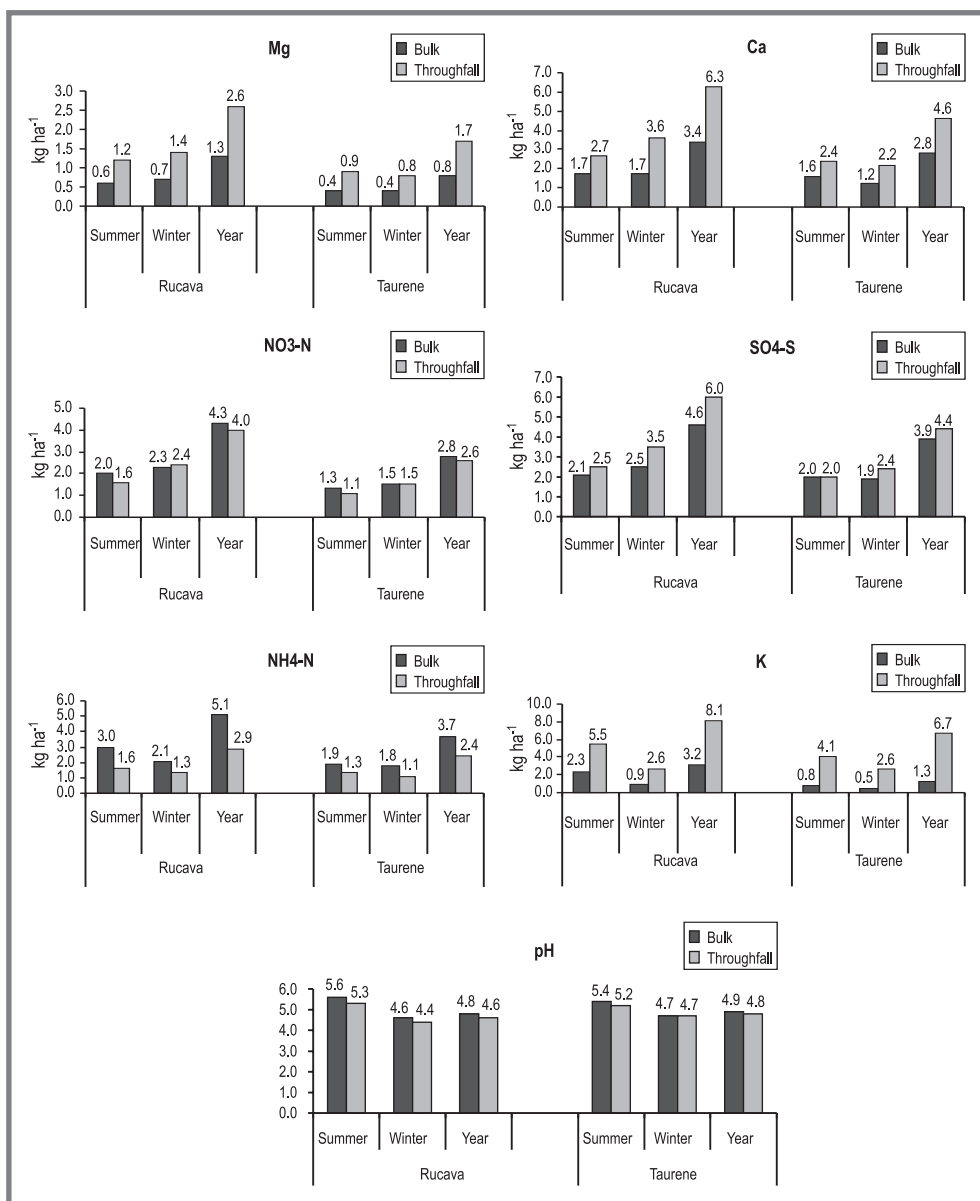


Figure 5.9 Mean annual amounts of deposits in bulk precipitation and throughfall in Rucava and Taurene in the 11 year period 1995-2005.

Mean amount of sulphur in Rucava was 4.6 kg ha<sup>-1</sup> a<sup>-1</sup> in bulk precipitation and 6.0 kg ha<sup>-1</sup> a<sup>-1</sup> in throughfall, in Taurene these values were 3.9 and 4.4 kg ha<sup>-1</sup> a<sup>-1</sup>, respectively. The amount of sulphur in precipitation was decreasing during the observation period but the trend was significant only for summer months (Table 5.6). The deposition of sulphur under the forest canopy has decreased by 19 % in Rucava and by 37 % in Taurene.

Mean amount of N in Rucava was 9.4 kg ha<sup>-1</sup> a<sup>-1</sup> (54 % in the form of ammonium) in bulk precipitation and 6.6 kg ha<sup>-1</sup> a<sup>-1</sup> in throughfall (48 % in the form of ammonium), in Taurene these values were 6.5 and 5.0 kg ha<sup>-1</sup> a<sup>-1</sup> (57 % and 48 % in the form of ammonium), respectively. For the amount of nitrogen there was a tendency to decrease during the observation period. The deposition of nitrogen under the forest canopy decreased by 88 % in Rucava and by 166 % in Taurene during the 11 year period.

Calcium, magnesium and potassium are important nutrition elements for plants. Mean annual amount of calcium in throughfall was 6.3 kg ha<sup>-1</sup> a<sup>-1</sup> in Rucava and 4.6 kg ha<sup>-1</sup> a<sup>-1</sup> in Taurene (increment in 11 years 111 and 78%, respectively); magnesium – 2.6 and 1.7 kg ha<sup>-1</sup> a<sup>-1</sup> (61 and 51%), and potassium – 8.1 and 6.7 kg ha<sup>-1</sup> a<sup>-1</sup> (81 and 337%). Following S. Smidt classification of deposits amounts (Smidt 1986), concentration of Ca and K in rain water is high, but concentration of Mg is low both in Rucava and Taurene. There is a positive trend line for Ca, Mg, and K amounts in throughfall both in summer and winter months. Increase of Mg and K in summer months and per year is statistically significant (Table 5.6).

Increase in the amount of cations and decrease of sulphur and nitrogen compounds are leading to slow but progressive neutralization of rain water. Rain waters of integrated monitoring stations are weakly acid. Mean annual pH of rain water was 4.6-4.8 in Rucava and 4.8-4.9 in Taurene. In summer months rain water has higher pH (Rucava – 5.3-5.6, Taurene – 5.2-5.4), but in winter months lower pH (Rucava – 4.4-4.6, Taurene – 4.7).

Dynamics of sulphur and nitrogen shows gradual decrease of deposit amounts. Similar trends are observed also at other integrated monitoring stations in Europe (Forsius *et al.* 2001). On the other hand, trend of cations is positive both in Rucava and Taurene. Possibly, the positive trend will remain also in the future due to intensive urbanisation. It should be emphasised, that amounts of sulphur, nitrogen and cations are higher in Rucava than in Taurene, although the amount of precipitation is very similar at both stations.

#### 5.3.6

#### Ellenberg indicator values

Changes in Ellenberg indicator values were very slight. There was no significant temporal trend in temperature. The only significant trend was for moisture (positive) in Peši site and for continentality (positive) in Taurene site. Brušviti site has been more dynamic as five from six Ellenberg indicator values have positive or negative significant trend (Table 5.7 & 5.8).

Table 5.7 Mean Ellenberg indicator values (calculated as mean from plots A, B and C) in three integrated monitoring sites.

Variable	Taurene					Peši					Brušviti				
	1994	1997	2000	2003	2006	1994	1997	2000	2003	2006	1994	1997	2000	2003	2006
Light	5.73	5.73	5.64	5.62	5.64	5.77	5.75	5.72	5.72	5.76	6.15	6.11	6.12	6.03	5.98
Temperature	3.24	3.28	3.21	3.10	3.11	3.75	3.73	3.75	3.71	3.86	3.38	3.54	3.41	3.42	3.46
Continentality	5.47	5.47	5.49	5.63	5.67	4.83	4.93	4.91	5.05	4.96	4.94	4.95	5.06	5.11	5.09
Moisture	4.21	4.27	4.31	4.26	4.22	4.45	4.46	4.55	4.54	4.56	4.23	4.29	4.24	4.22	4.21
Reaction	3.12	3.10	3.12	3.15	3.10	2.82	2.91	2.98	3.06	2.98	2.52	2.73	2.94	2.99	2.98
Nitrogen	2.07	1.97	2.05	2.09	2.11	2.54	2.60	2.69	2.68	2.65	1.74	1.81	1.85	1.93	2.00

Table 5.8 Trend (1994-2006) statistics for Ellenberg indicator values (Man-Kendal test).

Location	Ellenberg's value					
	Light	Temperature	Continentality	Moisture	Reaction	Nitrogen
Brušviti P	-1.96 0.025	0.98 0.163	1.95 0.25	-2.45 0.007	1.97 0.025	2.45 0.007
Peši P	-0.76 0.224	0.25 0.400	1.47 0.071	1.96 0.025	1.77 0.039	0.98 0.164
Taurene P	-1.57 0.059	-1.47 0.071	2.27 0.012	0.00 -	0.00 -	1.47 0.071

5.4

## Discussion

5.4.1

### Species diversity and frequency

Plant species diversity showed an increasing trend from 1994 to 2000 (2003 in Peši), but went down starting with 2003 (2006). Although the changes among years were statistically significant average differences in diversity parameters were very slight. The decrease of species richness in 2006 could be due to extremely dry year (the driest year in 85 years). It could also be that the time series is too short to reveal the real pattern of dynamics. Other studies have also shown that in a short time period species diversity can vary in boreal forests but without direction (Okland 1995; Liu, Bråkenhielm 1996; Grandin 2004).

Changes in species frequency correlate with the dynamics of Ellenberg indicator values. The most intensive changes in Ellenberg indicator values were observed in Brušviti, and this can be explained by the recent fire event.

All monitoring sites experience slow increase in nitrogen and reaction figure. Indicators of this process are the increase in frequency of *Vaccinium myrtillus*, *Scleropodium purum*, and *Cirriophyllum piliferum*, and the decrease in frequency of *Vaccinium vitis-idaea*, *Calluna vulgaris*, *Dicranum scoparium*. Common feature for all sites is also the decrease in light figure. The moisture figure shows increasing trend in Peši (such species as *Sphagnum capillifolium*, *Molinia caerulea*, and *Vaccinium uliginosum* increase) and decreasing trend in Brušviti, although these sites are only some kilometers from one another. Possible reason for drying of upper soil layer could be dense turf development of *Deschampsia flexuosa*. Such processes are observed in pine forests in Germany (Heinsdorf 1967).

Changes in species frequencies can be attributed both to eutrophication and also to natural aging of forest. The increase in *Maianthemum bifolium*, *Trientalis europaea*, *Hylocomium splendens*, and *Dicranum polystemum* with stand age is reported from the

southern Finnish pine forests (Nieppola 1992). The increase in *Scleropodium purum*, *Picea abies*, and *Vaccinium myrtillus* in NE Germany from 1965 to 1998 is explained as the effect of natural eutrophication process of stand development (Zerbes, Brande 2003). All the species mentioned (with the exception of *Dicranum polysetum* which has decreased significantly) have increased in Taurene and Rucava. *Deschampsia flexuosa* is often reported to be an indicator of nitrogen deposition (Heinsdorf 1967; Falkengren-Grerup 1989, 1990; Hofmann *et al.* 1990; Tamm 1991; Rodenkirchen 1993; Brunet *et al.* 1998; Bobbink *et al.* 1998; van Dobben *et al.* 1999), and this species has increased significantly in Brušviti site. On the other hand, *Dicranum polysetum*, *Chamaerion angustifolium*, *Rubus idaeus* and *Dryopteris carthusiana* are also the species reported as indicators of eutrophication (Mäkipää 1994; Bobbink *et al.* 1998) but in our case they do not show any significant increase in frequency.

#### 5.4.2

### Vegetation dynamics and environmental factors

Ordination revealed that there is a significant temporal gradient in vegetation dynamics during the 12 year period in pine forests. This gradient is more pronounced in Rucava than in Taurene and shows that forest vegetation changes from light and nutrient poor forests with low species diversity to more nutrient rich, species more diverse and closed forests with better moisture conditions. It is proved both by species abundance changes in the course of time and by changes of Ellenberg indicator value of nitrogen and reaction.

Tree and shrub layer is the least dynamic sinusia of forest community. The common feature is increase in abundance of *Picea abies* and decrease in *Pinus sylvestris*, and it is more pronounced in Taurene and Peši than in Brušviti. Ground layer experiences gradual replacement of *Calluna vulgaris* and *Vaccinium vitis-idaea* by *Deschampsia flexuosa* in Brušviti, by *Vaccinium myrtillus* in Taurene (*Deschampsia flexuosa* is not present there at all), and by both species in Peši.

However, it is hard to say if the temporal gradient of vegetation observed in this research is caused by air pollution, or if it is a consequence of natural vegetation dynamics. There is a very slight increase in Ellenberg nitrogen figures in the 12 years – from 0.04 degrees in Taurene to 0.26 degrees in Brušviti. This increment is even less than spatial differences in mean nitrogen figures among plots in the first observation year – mean nitrogen value vary (among plots A, B, and C) by 0.42 degrees in Brušviti, by 0.26 degrees in Peši and by 0.13 degrees in Taurene. Moreover, atmospheric nitrogen deposition that has been emphasized as the main factor for eutrophication in many studies (Falkengren-Grerup 1986; Brunet *et al.* 1998; Kuhn *et al.* 1987 etc.) has decreased both in Rucava and Taurene during the observation period. Eutrophication effect on boreal forest ground vegetation is reported from sites with about 15 to 40 kg ha<sup>-1</sup> a<sup>-1</sup> of atmospheric deposition of nitrogen. These figures are only 5-10 kg N ha<sup>-1</sup> a<sup>-1</sup> in Latvian monitoring sites. Some species do respond to such a low amount of deposition, for example *Deschampsia flexuosa* (Bobbink *et al.* 1998) but it seems that atmospheric nitrogen deposition in Latvian monitoring sites is still less than the critical load inducing substantial changes in forest vegetation. There are no precise data on critical nitrogen loads on boreal forests but it is assumed by expert judgment that it is about 10-15 kg N ha<sup>-1</sup> a<sup>-1</sup> (Bobbink *et al.* 1998).

On the other hand, increase in soil reaction is reported to have an indirect positive effect (improved biological activity of the humus) on eutrophication process (Becker *et al.* 1992). In both monitoring stations in Latvia increase in Ca and Mg ions in precipitation as well as increase in Ellenberg reaction value has been observed.

To interpret ordination results also forest spatial heterogeneity is an important aspect. In our data the first ordination axis correlated with factors governing both



spatial (inner plot heterogeneity in edaphic conditions) and temporal (changes among years) differences inside each site. It is suggested that the axis with the higher eigenvalue is the most important in revealing ecological gradients in data (Jongman *et al.* 1995). The strongest correlations with the first axis was for Ellenberg values of edaphic factors such as nitrogen and reaction and less tight correlation with the time scale. So we can conclude that inner vegetation heterogeneity in plots is more pronounced than the directed changes in the course of time.

Our results are in accordance with findings of other researchers. For example, Grandin (2004) has shown that with short time series (about 15 years) inner heterogeneity of vegetation plots can be of greater importance in explaining the variance in ordination analysis than temporal trend. Several authors **emphasize that** boreal forests are very heterogeneous in small spatial scales because of distinct micro relief and patchy structure of vegetation (e.g. Grandin 2004).

The pine forest ecosystems in both monitoring sites are still not mature. It means that natural vegetation dynamics towards mature pine forest is taking place. As the natural changes in vegetation in such forests leads to similar changes in flora and vegetation as eutrophication does (Becker *et al.* 1998), it is not possible to separate both influences in the present research.

There are regional differences in vegetation dynamics. More intensive eutrophication is going on in Rucava than in Taurene. It is obvious from both the vegetation ordination data as well as from dynamics of Ellenberg indicator values and atmospheric depositions. In Latvia this process is more intensive in uplands (Vidzeme, Rietumkursā, Alūksne etc.), where the final stage of oligomesotrophic pine forest succession is spruce forest (Laivīns 2005). In the Coastal Lowland invasion of spruce in pine forests is much slower, and spruce is more common only in field and shrub layer (only some individuals reach the tree layer).

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**Appendix I** Values of z-test ( $z_{\alpha} > z_{0.05} > 1.96$ ) for the differences in species frequency between 1994 and 2006, (+) species frequency increases, (-) species frequency decreases.

Species	Monitoring site		
	Brušviti	Peši	Taurene
<b>Trees and shrubs</b>			
<i>Betula pendula</i>	0.24 (+)	1.91 (-)	<b>2.71 (-)</b>
<i>Corylus avellana</i>	-	0.45 (-)	-
<i>Frangula alnus</i>	-	<b>2.52 (+)</b>	0.92 (-)
<i>Picea abies</i>	0.72 (+)	<b>3.68 (+)</b>	1.74 (+)
<i>Pinus sylvestris</i>	1.09 (-)	0.34 (-)	1.66 (-)
<i>Populus tremula</i>	-	1.15 (-)	1.01 (-)
<i>Quercus robur</i>	0.26 (+)	1.64 (+)	0.40 (-)
<i>Salix cinerea</i>	-	1.42 (-)	1.01 (-)
<i>Sorbus aucuparia</i>	-	1.63 (+)	0.79 (+)
<b>Herbs and dwarf shrubs</b>			
<i>Agrostis tenuis</i>	-	1.15 (-)	1.01 (-)
<i>Calamagrostis arundinacea</i>	-	1.25 (-)	-
<i>Calamagrostis epigeios</i>	-	1.75 (-)	-
<i>Calluna vulgaris</i>	<b>9.10 (-)</b>	<b>5.84 (-)</b>	<b>5.47 (-)</b>
<i>Carex ericetorum</i>	-	1.01 (-)	1.01 (-)
<i>Carex pilulifera</i>	-	0.00	-
<i>Chamaerion angustifolium</i>	-	0.34 (-)	1.01 (+)
<i>Convallaria majalis</i>	-	1.42 (+)	-
<i>Deschampsia flexuosa</i>	<b>2.71 (+)</b>	-	-
<i>Dryopteris carthusiana</i>	-	0.82 (-)	<b>2.26 (-)</b>
<i>Empetrum nigrum</i>	1.28 (-)	0.00	-
<i>Epipactis helleborine</i>	-	0.00	-
<i>Festuca ovina</i>	-	<b>3.52 (-)</b>	<b>4.27 (-)</b>
<i>Festuca rubra</i>	-	1.42 (+)	-
<i>Goodyera repens</i>	-	-	<b>8.32 (+)</b>
<i>Hieracium vulgatum</i>	-	0.45 (-)	-
<i>Lathyrus sylvestris</i>	-	0.45 (-)	-
<i>Luzula pilosa</i>	-	<b>5.57 (-)</b>	<b>5.78 (-)</b>
<i>Lycopodium annotinum</i>	-	1.01 (+)	1.42 (-)
<i>Lycopodium clavatum</i>	-	2.56 (-)	-
<i>Maianthemum bifolium</i>	-	<b>9.32 (+)</b>	0.51 (+)
<i>Melampyrum pratense</i>	<b>5.15 (+)</b>	1.28 (-)	<b>6.79 (-)</b>
<i>Molinia caerulea</i>	-	1.15 (+)	-
<i>Monotropa hypopitys</i>	1.42 (-)	1.01 (-)	-
<i>Orthilia secunda</i>	-	1.01 (-)	-
<i>Pteridium aquilinum</i>	-	<b>3.32 (+)</b>	0.00
<i>Pyrola chlorantha</i>	-	1.91 (+)	-
<i>Rubus idaeus</i>	-	1.01 (-)	-
<i>Rumex acetosella</i>	-	-	1.01 (+)
<i>Solidago virgaurea</i>	-	0.72 (-)	0.00
<i>Trientalis europea</i>	-	<b>8.37 (+)</b>	1.43 (-)
<i>Vaccinium myrtillus</i>	<b>6.90 (+)</b>	<b>5.04 (+)</b>	0.34 (+)
<i>Vaccinium uliginosum</i>	0.40 (+)	1.42 (+)	0.34 (-)
<i>Vaccinium vitis-idaea</i>	0.54 (-)	<b>7.16 (-)</b>	0.92 (-)
<i>Vicia cassubica</i>	-	1.01 (+)	-

<b>Mosses and lichens</b>			
<i>Aulacomium palustre</i>	<b>2.26 (+)</b>	<b>2.34 (-)</b>	0.58 (-)
<i>Brachythecium oedipodium</i>	<b>4.20 (+)</b>	<b>4.52 (+)</b>	<b>6.43 (+)</b>
<i>Cirriphillum piliferum</i>	<b>3.81 (+)</b>	<b>13.56 (+)</b>	<b>2.67 (+)</b>
<i>Cladina arbuscula</i>	<b>3.10 (-)</b>	-	-
<i>Cladina rangiferina</i>	0.16 (-)	1.01(-)	1.76 (-)
<i>Cladina stellaris</i>	<b>2.26 (-)</b>	-	-
<i>Dicranum polysetum</i>	0.00	<b>8.73 (-)</b>	<b>3.58 (-)</b>
<i>Dicranum scoparium</i>	<b>7.07 (-)</b>	<b>3.82 (-)</b>	<b>9.03 (-)</b>
<i>Hylocomium splendens</i>	<b>8.03 (+)</b>	1.83 (+)	0.00
<i>Hypnum jutlandicum</i>	<b>2.86 (+)</b>	-	-
<i>Lophocolea heterophylla</i>	-	-	1.01 (+)
<i>Pleurozium schreberi</i>	1.42 (+)	1.01 (-)	0.00
<i>Pohlia nutans</i>	-	-	1.01(-)
<i>Polytrichum formosum</i>	-	0.58 (-)	1.01 (+)
<i>Polytrichum juniperinum</i>	<b>3.04 (+)</b>	0.58 (+)	1.75(-)
<i>Ptilidium ciliare</i>	0.82 (-)	-	-
<i>Ptilium cirsta-castrensis</i>	1.75 (+)	<b>2.81 (-)</b>	<b>3.85 (+)</b>
<i>Rhytidiadelphus triquetrus</i>	-	1.01 (+)	1.01 (+)
<i>Scleropodium purum</i>	<b>13.58 (+)</b>	<b>6.40 (+)</b>	-
<i>Sphagnum capillifolium</i>	-	0.26 (+)	-
<i>Sphagnum girgensohnii</i>	-	-	1.01 (+)

## 6 Reports on national ICP IM activities

### 6.1

#### Report on national ICP IM activities in Austria

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The ICP IM programme is carried out at one site in Austria since 1992, the Zöbelboden (AT01) in the Northern Limestone Alps. In 2006 the focus was firstly on analysing long-term trends of bioindicators (Table 6.1.1) in relation to nitrogen and sulphur deposition. Secondly, a multi-isotope approach (N, S, O, Sr and Pb) was carried out to estimate the impact of long distance air pollution on the karst groundwater.

#### Effects of Nitrogen and Sulphur on bioindicators

The soils responded to the long-term trends in the deposition of airborne pollutants. Excess N deposition caused soil eutrophication and the decrease of S deposition resulted in a significant, but soil-specific, recovery from acidification.

The detected trends of soil properties were not unambiguously reflected in changes of forest floor vegetation. Though nitrophilous species increased in abundance, the overall change of the species composition of forest floor vegetation is only weak. However, the weak response predominates under intermediate site conditions, which may mask significant changes on oligotrophic sites.

Terrestrial and epiphytic bryophyte species remained rather stable in their overall abundance. The observed changes, which are restricted to single species, can only to some extent be attributed to effects of airborne N and S pollution. The bryophyte communities as a whole, however, did not show directional changes attributable to the observed amounts of N deposition and the decrease of S deposition. However, the spatial distribution of bryophyte communities is related to different deposition regimes of airborne N and S, which indicates long-term chronic effects.

Long-term airborne N and S deposition had a significant impact on epiphytic lichens. Though acid deposition through S decreased, epiphytic lichens did not recover. Additionally, lichens show clear signs of eutrophication due to excess N deposition. The overall abundance of epiphytic lichens and those of sensitive species decreased. Some sensitive species even became extinct. Lichen communities show a deteriorating development in response to air pollution and are becoming increasingly dissimilar from communities typical of clean air conditions.

Table 6.1.1 Overview of long-term data of the Austrian ICP IM site which was used for the present study. Years of survey are given as dark shaded boxes.

year of survey	1991 – 1999									2000 – 2005					
	1	2	3	4	5	6	7	8	9	0	1	2	3	4	5
forest inventory															
large herbivore pressure															
soil															
forest floor vegetation															
terrestrial and epiphytic bryophytes															
epiphytic lichens															
birds															

### Multi-isotope approach

Karst and other sensitive aquifers contribute up to 90 % to the total drinking water supply in some European regions. However, they are more vulnerable to contamination than other aquifers due to short transfer times from recharge to source. Therefore, the main objective of this study is to show possibilities to quantify the impact of even small long distance air pollution on water resources.

In a pilot study, precipitation, soil, rock and spring waters were collected. The hydrochemistry and the isotopic composition of nitrate, sulphate, strontium, lead and the water molecule itself has been analysed in five laboratories, each of them specialised in a certain group of isotopes.

Comparison of strontium isotope measurements in precipitation, spring waters and dolomite bedrock with literature data indicate that  $^{87}\text{Sr}/^{86}\text{Sr}$ -isotope ratios in precipitation (0.7092) support at least a more radiogenic, far transported source in addition to a possible recycling of local dolomite and limestone dust (0.7080-0.7083). Spring waters show similar ratios (0.7083-0.7084) confirming Sr-isotopes are good indicators for groundwater contact with specific host rocks.

The monthly precipitation samples show  $^{18}\text{O}$ -rich sulphate ions, whereas the soil sulphates change in a direction to lower  $^{18}\text{O}$ - and higher  $^{34}\text{S}$ -values with depth. The spring waters and the bedrock dolomites vary in the range of  $\delta^{34}\text{S}$ -values (4-9 ‰). Assuming the precipitation samples and the dolomite bedrocks are end members the straight contribution of atmospheric sulphate without biogenic cycling can be estimated to be 20 % in the spring waters and 10-45 % in the soil samples.

The monthly precipitation and total deposition samples show  $^{18}\text{O}$ -rich nitrate ions, whereas the spring waters show variable influence of soil nitrates. Assuming the field of soil nitrification and the precipitation as end members a direct atmospheric nitrate contribution of 10-30 % derived from fossil fuel burning and agricultural emissions can be calculated.

Radiogenic Australian gasoline-lead still dominates with 60-80 % the composition of the trace lead in the spring waters. In addition to the lead leached from the dolomite bedrock a third source contributes about 5-10 %. This second long distance Pb-contribution may originate from coal burning and/or Ag-Pb-ore smelting in Central Europe in the past.

### Publications

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- Kralik, M., Humer, F., Grath, J., Numi-Legat, J., Hanus-Ilmar, A., Halas, S. and Jelenc, M. 2006. Impact of long distance air pollution on sensitive karst groundwater resources estimated by means of Pb-, S-, O- and Sr-isotopes. In: Duran, J.J., Andreo, B. Y Carrasco, F. (eds.). Karst, cambio climático y aguas subterráneas. Publicaciones del Instituto Geológico y Minero de España. Serie: Hydrogeología y Aguas Subterráneas N.º 18. 311-317, Madrid.
- Zechmeister, H.G., Dirnböck, T., Hülber, K., Mirtl, M. 2006. Assessing airborne pollution effects on bryophytes - lessons learned through long-term integrated monitoring in Austria. Environmental Pollution (in press).



## Report on national ICP IM activities in Estonia

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ICP IM Programme has been carried out at two monitoring sites in Estonia since 1995. Vilsandi (EE01) area is located on Estonia's westernmost island (58°23' N, 21°50' E) and Saarejärve (EE02) is located at the forested sub-catchment area (109 ha) of Lake Saare in eastern Estonia (58°39' N, 26°45' E). Stand characteristics of permanent plots of the monitoring sites are presented in Table 6.2.1.

Reporting year 2006 was one of the driest monitoring years and, hence, it was characterized by lower open land, throughfall, soilwater and runoff water fluxes compared to the average annual water fluxes for the period 1995-2005 (Table 6.2.1, in brackets).

In 2006 sampling and measurements were carried out under the sub-programmes: AM, AC, PC, TF, SF, SC, SW, LF, FD, EP, MB at both areas, and, additionally, under RW and AL at Saarejärve (EE02) area.

Table 6.2.1 Stand characteristics and average annual water fluxes (mm) during 1995-2005 compared with 2006 water fluxes (in brackets) of permanent plots at Estonian integrated monitoring areas.

Permanent plot	Vilsandi (EE01) pine stand	Saarejärve (EE02) pine stand	Saarejärve (EE02) spruce stand
Site type	<i>Fragario-Pinetum</i>	<i>Rhodococco-vitis-idaeo-Pinetum</i>	<i>Vaccinio-myrtilli-Piceetum</i>
Soil type	<i>Calcari-Gleyic Leptosol</i>	<i>Haplic Podzol</i>	<i>Haplic Podzol</i>
Age of dominant trees	100	120	90
Precipitation	535 (510)	630 (488)	
Throughfall	282 (226)	513 (387)	430 (335)
Soilwater amount from depth of 10 cm	101 (78)	57 (43)	113 (89)
Soilwater amount from depth of 40 cm	98 (67)	25 (24)	74 (85)
Runoff		100 (80)	

### Bulk precipitation and throughfall chemistry in 2006

High concentrations of SO<sub>4</sub>-S in bulk precipitation (about 1.0 mg l<sup>-1</sup>) and in throughfall (1.6 and 1.8 mg l<sup>-1</sup> in pine and spruce stand, respectively) during the first quarter of the year in Saarejärve raised SO<sub>4</sub>-S annual deposition to the highest level of the last four years in the open area and pine stand. While the maximum decrease of SO<sub>4</sub>-S load on open area has been about four-fold from 12 (1996) to 2.3 kg ha<sup>-1</sup> (2004), in 2006 the load of SO<sub>4</sub>-S was 2.8 kg ha<sup>-1</sup>. Annual average concentrations of cations (Ca and Mg) in bulk precipitation and in throughfall also showed the highest values in the last four years. For example, extra high Ca concentrations in bulk precipitation were measured during the first three months of the year (up to 5.5 mg l<sup>-1</sup> in January),

but other average monthly concentrations were also mostly higher than 1 mg l<sup>-1</sup> and so raised the annual load of Ca up to 6.7 kg ha<sup>-1</sup>. Therefore, deposition by bulk precipitation and throughfall of SO<sub>4</sub>-S, Ca and Mg showed increased values in 2006 after a distinct decline since 2002 at Saarejärve (EE02) area.

In Vilsandi area high anion and cation concentrations were measured in July and August while the pH of deposition showed the highest values of the monitoring period. During summer months the increased ion concentrations in deposition were probably caused by numerous forest fires in Estonia and in the transboundary areas of Russia. Despite the high summer concentrations of some ions, downward trends of annual average SO<sub>4</sub>-S in bulk precipitation and in throughfall, and NO<sub>3</sub>-N concentrations in bulk precipitation continued in 2006 at Vilsand area. Statistically significant (p<0.1) increasing trend of NH<sub>4</sub>-N (1994-2006) is characteristic in pine throughfall on the remote island of Vilsandi. This could be caused by natural decomposition of seaweed, because there is no farming on the island.

#### **Al and SO<sub>4</sub>-S in soilwater**

High positive correlation (r=0.903) between sulphate in throughfall and topsoil water is characteristic especially for the pine stand, while it is weaker for the spruce stand at Saarejärve. This is partly due to the use of zero-tension lysimeters (fitted below the organic horizon, at about 10 cm and below the eluvial horizon (40 cm). Therefore, increased SO<sub>4</sub>-S concentrations in pine throughfall in 2006 resulted in increased SO<sub>4</sub>-S contents in soilwater especially below the organic horizon of both stands at Saarejärve.

In 2006 highest concentrations during the monitoring period (1995-2005) of the following elements were measured in soil water of Saarejärve: both total Al (up to 1.4 and 1.1 mg l<sup>-1</sup> below organic, and up to 2.0 and 2.8 mg l<sup>-1</sup> below eluvial horizon in the pine and spruce stands, respectively) and soluble free Al<sup>3+</sup> (up to 1.0 and 0.75 mg l<sup>-1</sup> below organic and up to 1.1 and 1.4 mg l<sup>-1</sup> below eluvial horizon in the pine and spruce stands, respectively). Statistically significant increasing time trends (estimated by Mann-Kendall nonparametric test) of total soluble Al in soilwater from both depths of the pine and spruce stands and of free Al<sup>3+</sup> in soil water from both horizon of the pine stand characterize the study period of 1995-2005.

Only additional H<sup>+</sup> can increase both total Al and soluble free Al<sup>3+</sup> concentration in soil water (Figure 6.2.1). **The pH of solutions in the studied soil water was normally well below 5 at Saarejärve and, in fact, the pH of throughfall was commonly higher than that of the soil solution receiving it.** The increase of total soluble Al in podzolised soil indicates an ongoing process of podzolisation due to dissociated organic acids derived from mineralisation of conifer litter, and is probably attributable to decreased deposition of cations and decreased retention of sulphate in soil horizons (Frey, *et al.*, 2006).

The same tendency of statistically significant decrease of pH and SO<sub>4</sub>-S with some increase of total soluble Al concentrations (not significant) is characteristic to soilwater below the organic horizon (about 10 cm), but not for the deeper layer in Vilsandi pine stand.

#### **High SO<sub>4</sub>-S concentrations in runoff water**

On average 100 mm surface water a year runs from I sub-catchment of Lake Saare through a streamlet into the lake. This constitutes about 19% of the annual precipitation. In the dry year of 2006 the runoff flux was 80 mm (Table 6.2.1), the flowbed stayed dry in July, August and September. High SO<sub>4</sub>-S concentration (16 mg l<sup>-1</sup>) was measured after restoration of flow in October and also in November (13.8 mg l<sup>-1</sup>). Sharp increases of SO<sub>4</sub>-S concentrations were measured also after a dry and hot summer in 2002. A net release of SO<sub>4</sub>-S (3.1 kg ha<sup>-1</sup>) from catchment occurred due to S-mineralization

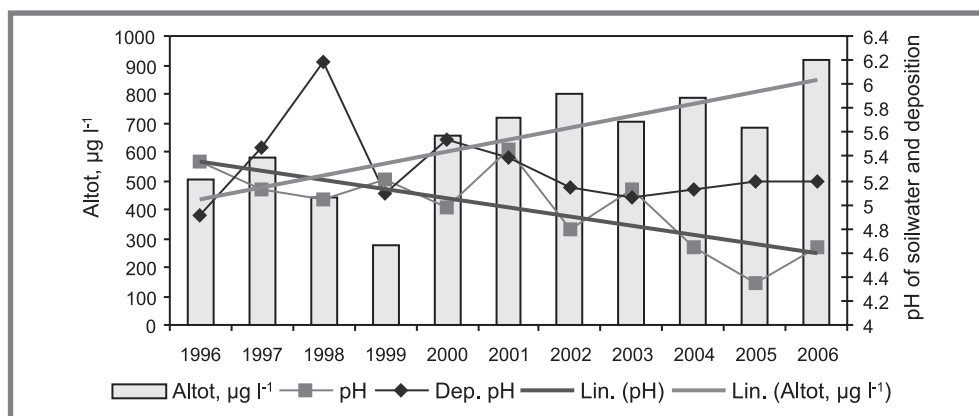


Figure 6.2.1 Increasing trend of annual average total soluble Al concentrations ( $\mu\text{g l}^{-1}$ ) and decreasing trend of pH in soilwater under organic horizon of the pine stand at Saarejärve during 1997-2006.

processes in the catchment and/or oxidation of previously retained reduced form of sulphur during drying of bottom sediments in the flow bed.

### Heavy metal input by litterfall and storage in soil organic horizon

Concentrations of heavy metals such as Zn, Cd, Pb and Cu in litter needles (estimated from an annual compositing sample) during 1995-2006 show statistically significant increasing trends for Cu and Cd in Vilsandi. This is, however, not a clear evidence of an increased external atmospheric input, because litterfall includes uptake from the soil, and Cu and Cd concentrations are increased in the current year needles, too, but not significantly. At the same time increased Cu and Cd concentrations in needles and litter needles could indicate increased mobility of these elements in the soil organic horizon and promoted uptake by pine roots. The store of Cu in upper organic layer (0-5 cm) in Vilsandi pine stand is high (Table 6.2.2), compared with data from Saarejärve and from ICP Forest database (Rademacher, 2001).

Heavy metal (Pb, Cd, Cu and Zn) concentrations ( $\text{mg kg}^{-1}$ ) in soil organic layer (estimated from two depths: 0-5 cm and 5-10 cm) of the studied stands at Saarejärve and Vilsandi (Table 6.2.2) show elevated Pb concentrations in both stands at Saarejärve and elevated Cu concentration in the top organic layer in Vilsandi pine stand and, to a certain extent, in Saarejärve pine stand. Zn-concentrations are elevated in all stands, especially in top organic. Lower concentrations of Pb in top organic horizon compared with deeper organic layer indicate probably decreased atmospheric input, assuming that uptake of the least mobile Pb by vegetation is negligible.

Table 6.2.2 Heavy metal concentrations ( $\text{mg kg}^{-1}$ ) in organic layer of Saarejärve spruce and pine stands in 2006 and Vilsandi pine stand in 2005.

Stand and depth of oranic layer	Heavy metal concentrations, $\text{mg kg}^{-1}$			
	Pb	Cd	Cu	Zn
Spruce stand, 0-5 cm	20	0.35	4.6	43
Spruce stand, 5-10 cm	48	0.34	3.8	28
Pine stand at Saarejärve, 0-5 cm	19	0.4	6.1	58
Pine stand at Saarejärve, 5-10 cm	59	0.37	5.2	47
Pine stand at Vilsandi, 0-5 cm	8	0.25	14	65
Pine stand at Vilsandi, 5-10 cm	10	0.25	2.0	23
Background (ICP Forests)	15	0.35	5	35
Elevated (ICP Forests)	15-150	0.35-3.5	5-20	35-300

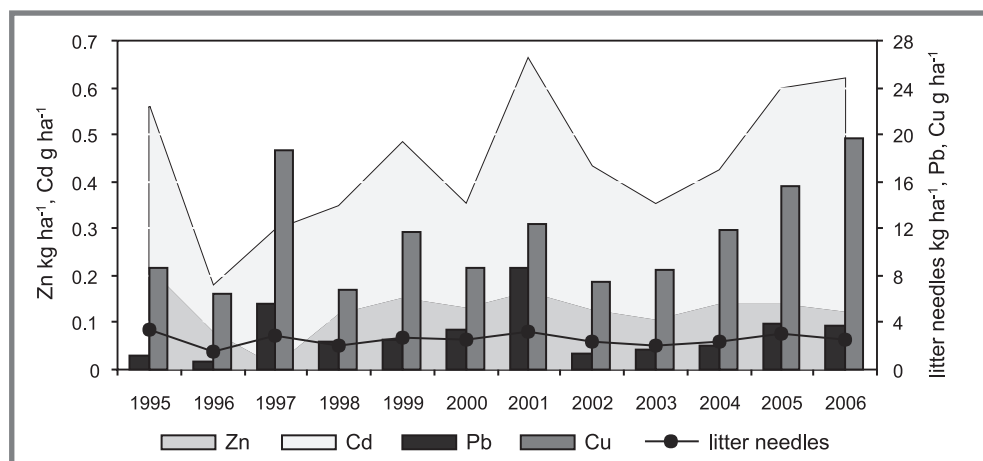


Figure 6.2.2 Input of Zn, Cd, Pb and Cu (g ha<sup>-1</sup>) by litterfall (kg ha<sup>-1</sup>) into Vilsandi pine stand during 1995-2006.

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## Report on national ICP IM activities in Germany

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As a contribution to the this year's 15<sup>th</sup> Integrated Monitoring Task Force Meeting at Grafenau (Germany) Bavarian Forest National Park Administration and German Federal Environment Agency jointly prepared a field trip to the IM site Forellenbach (DE01).

During the walking-tour to the measurement sites and the monitored tree stands, site characteristics and some results of long-term monitoring of environmental impacts and ecosystem changes were presented.

### Fifteen years of monitoring in the Forellenbach area – using mass balances, bioindication, and modelling approaches to detect air pollution effects in a rapidly changing ecosystem: main results

#### 6.3.1

#### Site description

The Forellenbach experimental area (0.69 km<sup>2</sup>) is a representative transect through the main forest types of this part of the Bavarian Forest. This transect is 2.9 km long with elevations between 787 and 1292 m. The mean elevation and inclination are 870 m and 12 %; 69 % of the area has a moderate inclination (3 – 8 %).

The bedrock consists of coarse granite (Älterer Finsterauer Kristallgranit) of the Carboniferous age. The lower elevations of the study area (58 %) are covered by sandy-loamy Dystric and Podzolic Cambisols 60 – 100 cm deep on periglacial deposits. At higher elevations, in the more eroded parts of the study area, these soils are associated with Rankers and Lithosols. Hydromorphic soils (30 % of the area) are found around springs and on solifluction sediments whose bottom layers are compacted. Mineral soils are acidified, they have a pH<sub>KCl</sub> of 3.1 in the topsoil layers and pH<sub>KCl</sub> 4.0 at 45 cm depth and show low percent of base saturation and high degree of Al saturation (< 10 % and 80 – 95 %, respectively).

High annual precipitation rates, abundant snow cover, and low mean annual temperatures are characteristics of the climatic conditions (1991 – 2006): areal precipitation was 1582 mm p.a., mean air temperature was 6.2°C. Between 800 and 1300 m a.s.l. annual precipitation increases with elevation by + 100 mm and temperature decreases by – 0.56 K / 100 m. In 1990, more than 95 % of the Forellenbach area was covered by forests. The stand was dominated by about 100-year-old *Picea abies* (L.) (Norway spruce, 69 %) and about 70-year-old *Fagus sylvatica* (L.) (European beech) with small amounts of *Acer pseudo-platanus* (L.) (sycamore maple) and other deciduous tree species. According to the climatic conditions, spruce stands cover the high and low elevations, while beech stands and mixed stands cover the slopes.

By 2003, the total area share of the spruce stands (Figure 6.3.1) fell to about 30 % by bark beetle attack (*Ips typographus* L.). Between 1995 and 2001, the mean volume stock of timber over bark of living spruce decreased from 478 m<sup>3</sup> ha<sup>-1</sup> to 206 m<sup>3</sup> ha<sup>-1</sup>. At high elevations more than 99 % of spruce trees had died (- 473 m<sup>3</sup> ha<sup>-1</sup>) while the lower elevations were less affected (Figure 6.3.2). This rapid change in the vegetation cover caused dramatic biogeochemical changes on plot scale as well as on catchment scale, which interact with changes in deposition.

Figure 6.3.1 Share of beech, spruce, and dead spruce stands in the Forellenbach area.

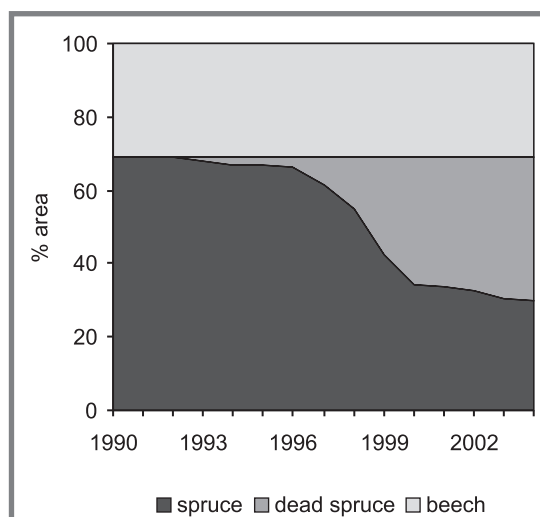
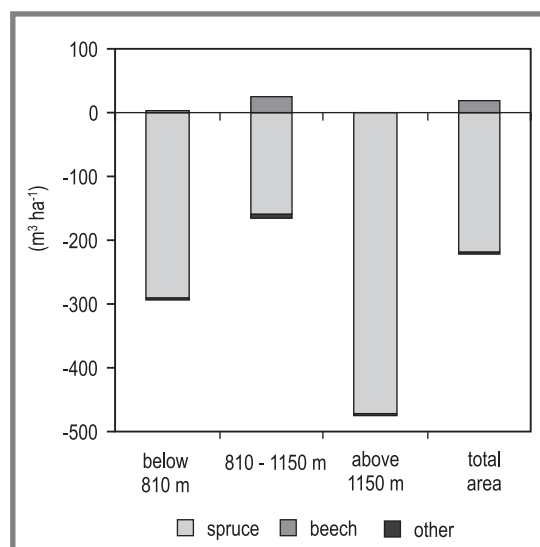


Figure 6.3.2 Changes in timber volume over bark ( $\text{m}^3 \text{ha}^{-1}$ ) between 1995 and 2001.



### Immission and deposition

Measurements of immissions and of meteorological parameters (temperature, humidity, insolation, wind) are carried out 50 m above ground at the Schachtenau tower (807 m a.s.l.). Concentrations of  $\text{SO}_2$  and  $\text{NO}_2$  in ambient air are relatively low at this elevation. From 1991 to 2004 annual means varied between 2 and 4  $\mu\text{g m}^{-3}$  for  $\text{SO}_2$  and between 4 and 9  $\mu\text{g m}^{-3}$  for  $\text{NO}_2$ , with no clear trend. Investigations into  $\text{NH}_3$  in ambient air showed low monthly mean concentrations (0.2 – 2.1  $\mu\text{g m}^{-3}$ ). The yellowing of spruce needles, which was a common phenomenon in this area in springtime and which was attributed to acidifying gases, disappeared to a large extent in lower elevations. Short-term and long-term concentrations of these noxious gases are well below the critical levels.

With respect to human health and tree vitality, ozone is the most important gaseous pollutant in this region (Figure 6.3.3). Annual means varied between 53 and 85  $\mu\text{g m}^{-3}$ , maximum daily and hourly means reached 180 and 240  $\mu\text{g m}^{-3}$  in the early 1990s, respectively. While peak concentrations have decreased, annual mean concentrations have increased by about 1  $\mu\text{g a}^{-1}$  since 1991. Ozone concentrations go up with increasing elevation by about 3  $\mu\text{g}$  per 100 m elevation.

Since the 1980s, measures to reduce  $\text{SO}_2$  emissions have reduced S deposition on the beech stands B1 (see below) by about 75 % (Figure 6.3.4) and even to a higher degree on spruce stands. Total N input (determined according to Ulrich's procedure)



decreased from more than  $20 \text{ kg ha}^{-1} \text{ a}^{-1}$  to annual rates between 10 and  $15 \text{ kg ha}^{-1}$  at this low elevation stand.

However, N input has increased substantially according to elevation (Figure 6.3.5), due to canopy interception of particles and gases, especially in spruce stands. These high loads correspond to modelled deposition loads for this region (Gauger *et al.* 2002). Even with today's reduced N emissions ( $\sim 25 \%$  since 1990), living high-elevation stands get about twice the N input of low elevation stands.

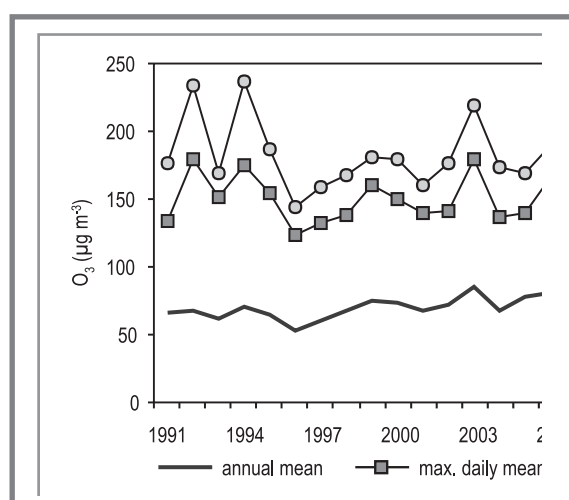


Figure 6.3.3 Ozone concentrations at Schachtenau measuring tower ( $\mu\text{g m}^{-3}$ ).

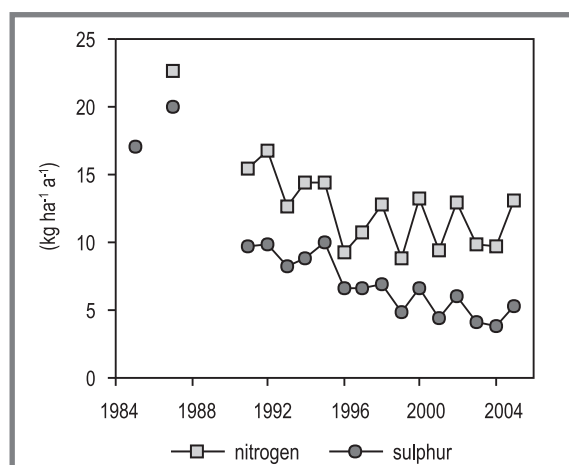


Figure 6.3.4 Total deposition of in-organic N and  $\text{SO}_4\text{-S}$  on beech stand BI Schachtenau ( $\text{kg ha}^{-1}$ ).

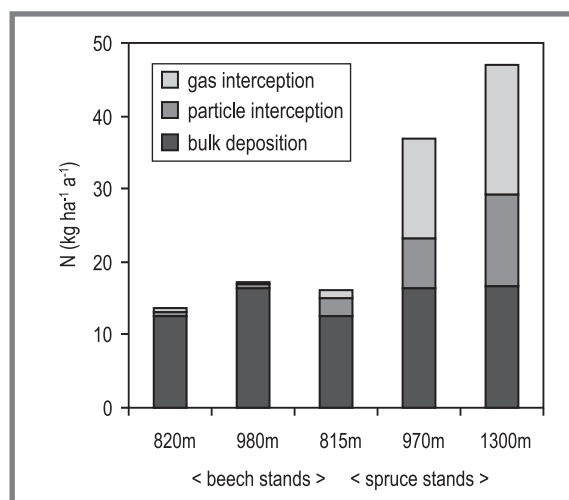


Figure 6.3.5 Total deposition and its components at different elevations (1991 – 1994).

### Climate change and changes in tree phenology

Since 1972 German Meteorological Service and National Park Administration have jointly operated the meteo station “Waldhäuser” at 945 m a.s.l., comprising synoptic weather observations and records of driving climate variables three times a day.

While precipitation has not shown any trend in monthly, seasonal and annual totals, air temperature has increased significantly with a mean annual increment of  $0.04 \text{ K a}^{-1}$  (Figure 6.3.6). This warming occurs from April to August. It is more affected by the increase in maximum than in minimum temperatures (Figure 6.3.7).

These thermal changes early in the year have affected the early phenological stages of tree species. The date of leaf unfolding of European beech shows a significant negative trend of about five to six days per decade (Figure 6.3.8), irrespective of the provenance used in the International Phenological Garden (IPG) near the meteo station Waldhäuser. Other broadleaves (poplar, birch, mountain ash) and Norway spruce (bud burst) showed significant advancements, too. For mountain ash the advancement increases with altitude. As the onset of senescence (yellowing) shifted to the same extent and direction, the length of vegetation period remained unaltered. A comparison with phenological observations in mature beech stands showed, that the date of leaf unfolding and its year-to-year variability is nearly identical to young beech clones at the same elevation and exposition.

Models to predict the date of bud burst and leaf unfolding from temperature alone (Kramer 1996, Menzel 1997, von Wilpert 1990) were examined by the use of these datasets. The Menzel-model satisfactorily mirrors the bud burst of Norway spruce without any adaption. For European beech, averaging the results of the von Wilpert- and the Kramer-model consistently reproduced the observed dates, after adding snow cover criteria and optimizing model parameters in each case.

Figure 6.3.6 Annual precipitation sum (mm) and mean air temperature ( $^{\circ}\text{C}$ ) at Waldhäuser meteo station since 1973 (hydrological years).

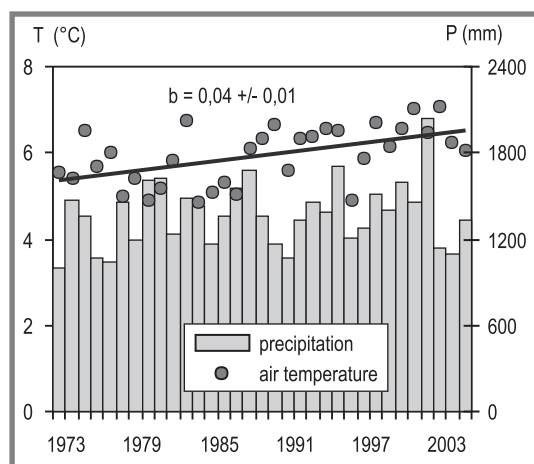
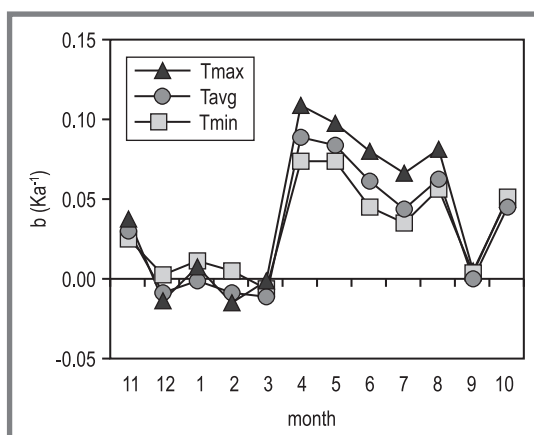


Figure 6.3.7 Monthly trend functions ( $\text{K a}^{-1}$ ) of mean air temperature and mean temperature extremes at Waldhäuser meteo station (1972 – 2006).



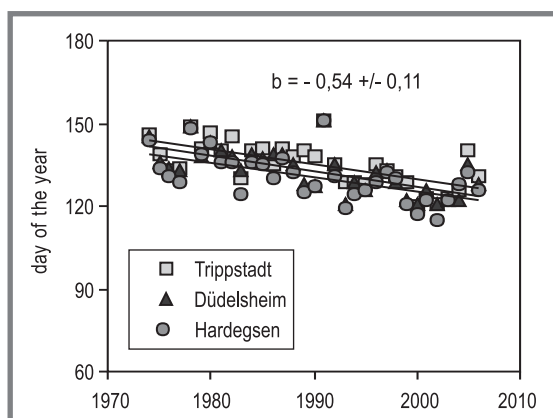


Figure 6.3.8 Date of leaf unfolding of three provenances of European beech at Waldhäuser International Phenological Garden.

### 6.3.2

## Ecosystem changes driven by air pollutants

In 2001, the beech stand B1 (about 95 years old, elevation of 820 m) had a timber volume over bark of 344 m<sup>3</sup> ha<sup>-1</sup>. The volume growth increment was 8 m<sup>3</sup> ha<sup>-1</sup> a<sup>-1</sup> between 1990 and 2001. The stand grows on a Dystric Cambisol, tending in some places to Podsollic Cambisol, which is acidified and low in exchangeable Mg<sup>2+</sup> and K<sup>+</sup>. This low supply is indicated by low or even insufficient contents of these ions in the leaves (see below). The cycling of water and solutes (precipitation, soil water, groundwater) and of biomass and organic bound components (stem growth, foliage, litter and litter decomposition) are monitored on this plot. Soil water contents in five depths (10, 30, 55, 85, and 115 cm) are measured in five pits each, using time domain reflectometry. Additionally, groundwater table is recorded half-hourly in the nearby well by a pressure sensor. On this plot, which is not directly affected by bark beetle damages, the biogeochemical changes are monitored, which are driven by changes in immission and deposition.

### Recovery from acidification and ongoing accumulation of nitrogen

In response to the marked downward trend in acid deposition, the soil water chemistry has been recovering (Figure 6.3.9): acid neutralizing capacity (ANC) has increased significantly at 100 cm (and 40 cm) depth, reducing concentrations of harmful aluminium ions. The increase in ANC was solely the result of decreasing sulphate concentrations; base cations remained at very low concentrations, which cannot be further reduced by uptake.

Since about 1994, nitrate concentrations in seepage water have been low (< 1 mg NO<sub>3</sub> l<sup>-1</sup>) throughout the mineral soil. Therefore N losses by seepage water (1 ± 2 kg N ha<sup>-1</sup> a<sup>-1</sup>) are small in spite of large amounts of precipitation, indicating that efficient sinks are currently existing in this ecosystem. The balance of total deposition and output by seepage water (1992 – 2003) reveals a net retention of 11 (± 2) kg N ha<sup>-1</sup> a<sup>-1</sup> (Figure 6.3.10).

The analysis of the internal N cycle shows, that the net uptake in harvestable biomass (stem wood and bark) is the most important sink for deposited N in the long term. The measured current net accumulation rate of about 9 kg N ha<sup>-1</sup> a<sup>-1</sup>, however, is about 50 % higher than the rate assumed for calculation of critical loads, due to comparatively high N contents in stem wood (1.25 g N kg<sup>-1</sup>). When neglecting small losses of gaseous N from this well drained soil, the net immobilization in soil organic matter adds up to 2 kg N ha<sup>-1</sup> a<sup>-1</sup>. Current N deposition rates would be slightly above the calculated critical loads (10 kg N ha<sup>-1</sup> a<sup>-1</sup>), assuming harvesting. But under National Park conditions (no management, no harvest) the N deposition is currently well above the acceptable deposition rate of about 4 kg N ha<sup>-1</sup> a<sup>-1</sup>.

Net mineralization of organic N is the quantitatively most important process in the N cycle. The net mineralization potential was estimated to about  $200 \text{ kg N ha}^{-1} \text{ a}^{-1}$  by using different methods. This amount alone should meet most of the demand of the trees in competition with micro organisms and understorey vegetation. For this reason, N storage in the soil has to be considered as the main process, currently preventing nitrogen leaching.

Nitrogen in the biomass of stand B1 amounts to  $709 \text{ kg N ha}^{-1}$  (Figure 6.3.11), almost equally divided into timber over bark and other compartments. But this comes up to only 6 % of the total available nitrogen pool of about  $11095 \text{ kg N ha}^{-1}$ . Huge amounts ( $10378 \text{ kg N ha}^{-1} \cong 94 \%$ ) are stored in the soil down to the lower boundary of the ecosystem (defined to be at 100 cm depth), and about half of this is stored in the upper mineral soil layers (0 – 40 cm depth). For this reason, the high mineralization rates (see above (s. a.)) appear very realistic.

The system's future capacity to immobilize deposited N in soil organic matter is unknown. C/N ratios of 20 – 22 in the organic layer and 16 – 18 in mineral soil indicate, that at present the narrowing of the C/N ratio is probably the key process for nitrogen storage.

This presents an increasing risk of nitrate leaching into groundwater, chronically as well as after abrupt disturbances like windthrow or insect pests (see below), which impair the system's producer – decomposer relations.

Figure 6.3.9 Annual median concentrations of  $\text{Al}^{3+}$  and acid neutralizing capacity (ANC) in seepage water at 100 cm depth.

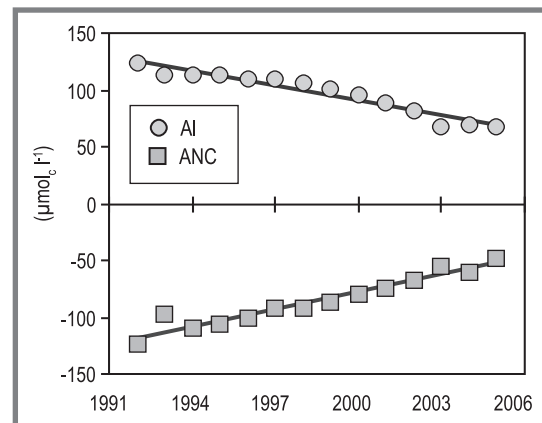
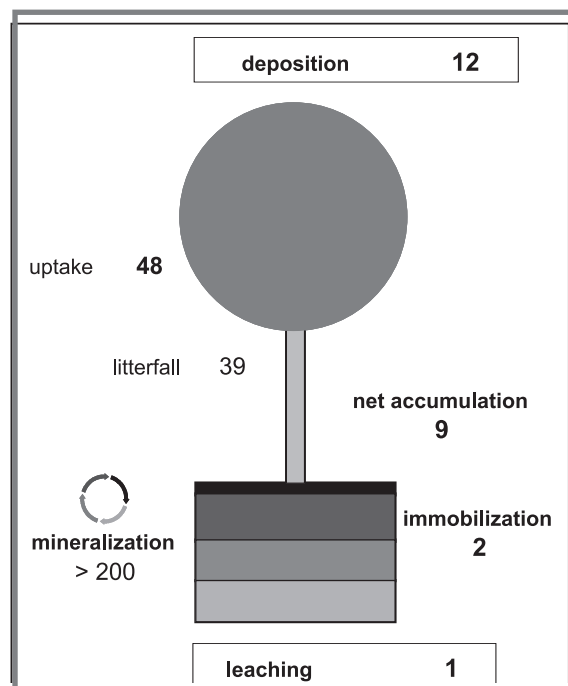


Figure 6.3.10 Simplified N cycle ( $\text{kg N ha}^{-1} \text{ a}^{-1}$ ) of the B1 beech plot.



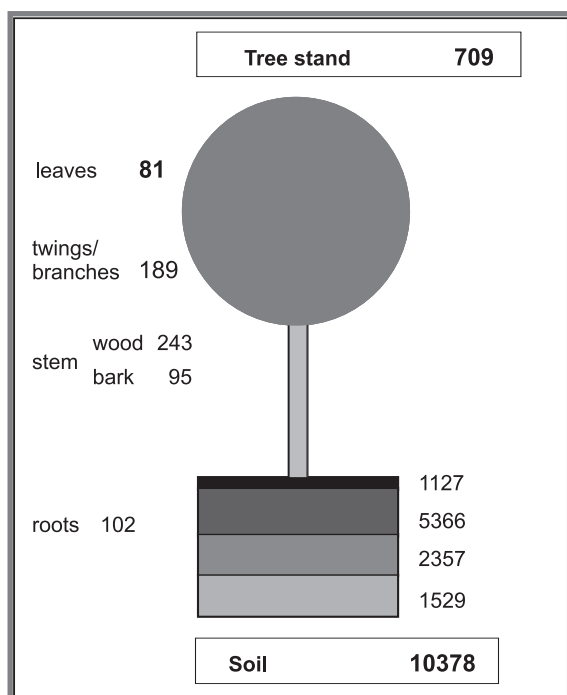


Figure 6.3.II Distribution of nitrogen (kg ha<sup>-1</sup>) in biomass (2001) and soil (1990) of beech stand B1.

### Effects of nitrogen on nutrition and growth

Tree growth of the beech stand B1 was evaluated in comparison to the same-aged beech stand B2 at higher elevations (980 m a.s.l.), which holds higher volume stocks as well as presently higher annual increments. The soil chemistry is very similar, but differs in a higher K<sup>+</sup> supply at B2, which is reflected by significantly higher contents in leaves. In contrast Mg<sup>2+</sup> contents in leaves are significantly lower in B2 than in B1, while N contents are higher, probably due to higher N deposition. In conclusion, nitrogen stimulates growth provided that the K<sup>+</sup> supply is sufficient; the Mg content may be reduced by high nitrogen uptake to near deficiency until becoming growth limiting.

In 1998, part of the B1 beech stand was influenced by bark beetle, which killed neighbouring spruce. Up to now, the affected beech trees have shown diameter increments of more than 100 % higher than before and more than those of the nonaffected beech, indicating a better nutrient supply. In fact, nutrient contents and nutrient relations in the leaves changed significantly (Table 6.3.1): K increased to sufficient values, N increased from 20 to 24 mg g<sup>-1</sup> dry matter and Mg decreased to rather insufficient values, but leaf weight and carbon content remained unchanged. N and Mg contents and relations became similar to the situation at stand B2, which are far above the balanced values (N/Mg: 12 – 25). This may destabilize the vitality of the trees by increasing sensitivity to drought, insect pests and pathogens.

With a view to groundwater protection, these findings suggest, that the tree stand's potential for additional growth and N accretion in biomass is restricted by such nutrient (base cation) limited site conditions.

Table 6.3.1. Nutrient contents and relations in beech leaves at stand B1 and B2 (1999 – 2001). (a): affected by spruce dieback

	B1	B1 (a)	B2
N mg g <sup>-1</sup>	20	24	24
K mg g <sup>-1</sup>	4.5	5.5	8.3
Mg mg g <sup>-1</sup>	0.83	0.67	0.64
N/K mg mg <sup>-1</sup>	5	5	3
N/Mg mg mg <sup>-1</sup>	25	38	37

### Assessment of vitality risks for beech stands by ambient ozone

A big-leaf model (ICP M&M Manual 2004) was run to estimate ozone deposition into the beech stand B1 and the relevant ozone concentrations at the top of its canopy. Input data were records from continuous measurements of ozone concentrations in ambient air and meteorological variables at 51 m above ground level (807 m a.s.l.) on Schachtenau measuring tower. A comparison with directly measured ozone concentrations at the top of the canopy (~25 m, three replicates) showed very realistic results for daylight hours from June to August 2006 (Figure 6.3.12). Varying the impact height in the model from 25 m (canopy height) to 23 m (0.5 m above the conceptual sink), which covers the measuring heights, resulted in a 1:1 slope of the regression lines with a 90 % coefficient of determination.

In a second step ozone risk indicators were calculated with this 3-month data set. The cumulative ozone exposure (AOT40) of sunlit leaves (and cumulative stomatal ozone uptake,  $AF_{ST,1,6}$ ) didn't show significant differences in monthly or seasonal values, when using simulated and measured ozone concentrations respectively (Figure 6.3.13). Apart from this comparisons, ozone exposition (and uptake) during two months in this summer was much higher than the critical level for the whole vegetation period.

This successfully parameterized and tested model was applied to the data sets of former years (Figure 6.3.14). Cumulative exposure (AOT40) and cumulative stomatal uptake ( $AF_{ST,1,6}$ ) indicate chronic and severe exceedances of the particular critical levels. The concentration based ( $Cle_c$ ) and the flux based critical level ( $Cle_f$ ) are provisionally set at 5 ppm h and 4 mmol  $O_3$  m<sup>-2</sup> PLA (projected leaf area).

The course of AOT40 closely followed the ozone concentrations with peak values in 2003. In contrast, limited water supply in that warm and dry summer reduced the stomatal gas exchange and thus  $AF_{ST,1,6}$  reached a minimum in this year.

Figure 6.3.12 Comparison of modelled and measured ozone concentrations ( $\mu\text{g m}^{-3}$ ) at the top of the canopy on beech stand B1.

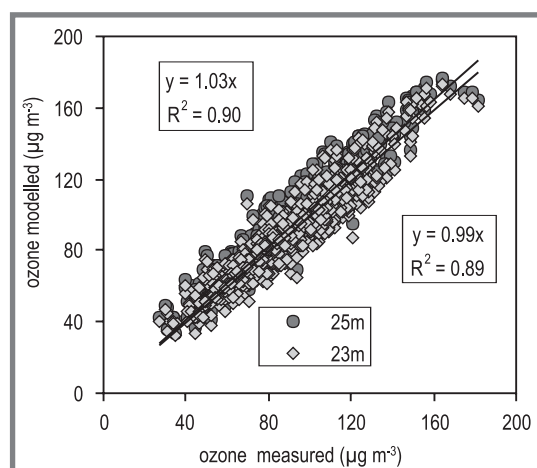
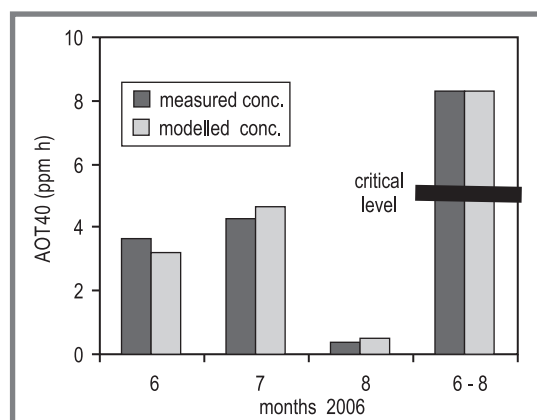


Figure 6.3.13 Cumulative ozone exposure (AOT40) calculated from measured and modelled ozone concentrations at canopy height.





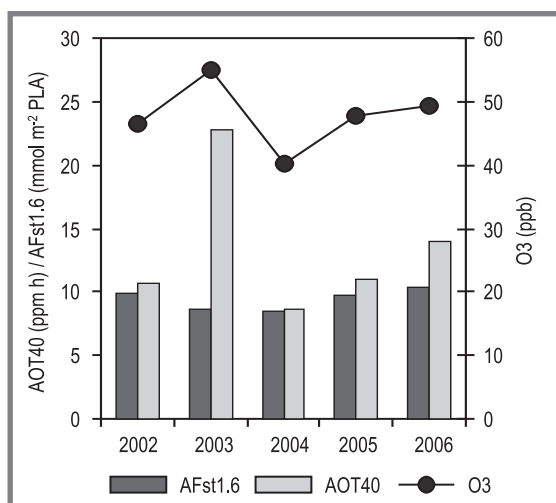


Figure 6.3.14 Cumulative ozone exposure (AOT40) and cumulative ozone uptake (AF<sub>st</sub>1.6) calculated from modelled ozone concentrations at canopy height.

### 6.3.3

## Ecosystem changes driven by insect pest

The spruce stand F1 (about 110 years old, at 815 m elevation) grew on a Dystric Cambisol, tending in some places to Podsollic Cambisol, which is similar to the soil at beech plot B1, but less poor in exchangeable cations. The timber volume over bark was 998 m<sup>3</sup> ha<sup>-1</sup> in 1995 and the volume growth increment was 16 m<sup>3</sup> ha<sup>-1</sup> a<sup>-1</sup> (1990 – 1995). Beech in the third tree layer (n = 240) contributed about 28 m<sup>3</sup> ha<sup>-1</sup> to the total volume stock. Nitrogen in the stand biomass amounted to 1303 kg N ha<sup>-1</sup> (Figure 6.3.15), but 89 % of the total nitrogen pool is stored in the soil (10524 kg N ha<sup>-1</sup>), predominantly in the upper mineral soil layers (0 – 40 cm depth). The potential net mineralization of organic N was estimated to about 200 kg N ha<sup>-1</sup> a<sup>-1</sup>. From June 1996 to the autumn of 1997, more than 99 % of spruce died after bark beetle attack. In 1999, most of the dead standing trees had fallen or broken down.

The cycling of water and solutes (precipitation, soil water, groundwater) and of biomass and organic bound components (stem growth, foliage, litter, and litter decomposition) has been monitored on this plot like on the adjacent beech plot B1 since 1990. This enables quantitative estimates of the consequences in water and element cycles, which are caused by these abrupt changes in the stand's structure, the micro-climatic conditions, and in the relations between producers and decomposers.

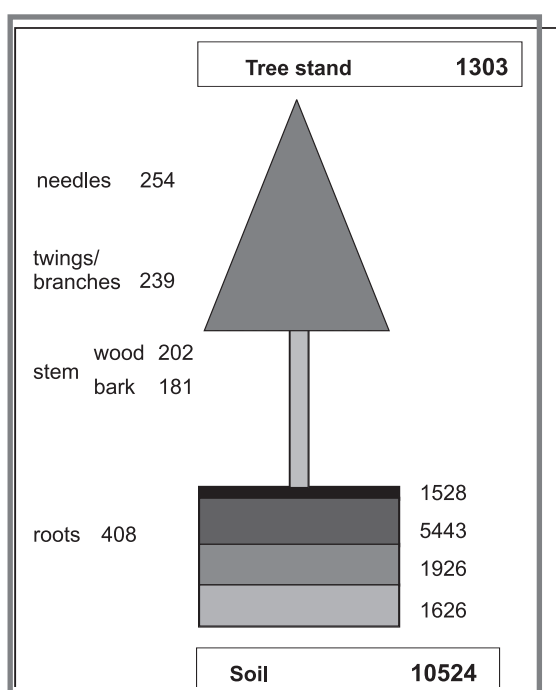


Figure 6.3.15 Nitrogen (kg ha<sup>-1</sup>) in biomass (1995) and soil (1990) of the spruce stand F1.

### Excess mineralization and nitrification

Already two months after the bark beetle attack, N concentrations began to increase in the organic layer percolate (Figure 6.3.16). Between 1997 and 2000 maximum concentrations in seepage water were 21 mg l<sup>-1</sup>, 43 mg l<sup>-1</sup>, and 28 mg l<sup>-1</sup> at 0 cm, 40 cm, and 100 cm depth respectively. In mineral soil waters, nitrate accounted for more than 97 % of total inorganic N. The difference in N concentrations at 40 cm and 100 cm depth may be due to a renewed incorporation of NO<sub>3</sub>-N into humus, since unchanged DOC and very small NO<sub>2</sub><sup>-</sup> concentrations indicate only small gaseous losses.

This excess production of nitric acid caused a maximum decrease of solution pH from 4.0 to 3.6 in organic layer percolate and from 4.7 to 4.0 in soil water at 40 cm depth. Al<sup>3+</sup> concentrations increased synchronically and stoichiometrically to NO<sub>3</sub><sup>-</sup> concentrations from less than 2 mg l<sup>-1</sup> to a maximum of 27 mg l<sup>-1</sup> (40 cm depth). These aluminium ions were released from exchange sites, which enabled this fast reaction. In contrast, SO<sub>4</sub><sup>2-</sup> concentrations decreased by about 70 %, because aluminium ions and protons controlled the solution of aluminosulphate minerals.

Since 2002, pH-values and ion concentrations in seepage water have nearly reached previous levels, suggesting, that the reserves of readily degradable organic nitrogen in fresh litter (green needles, fine roots) and in soil organic matter has now been exhausted.

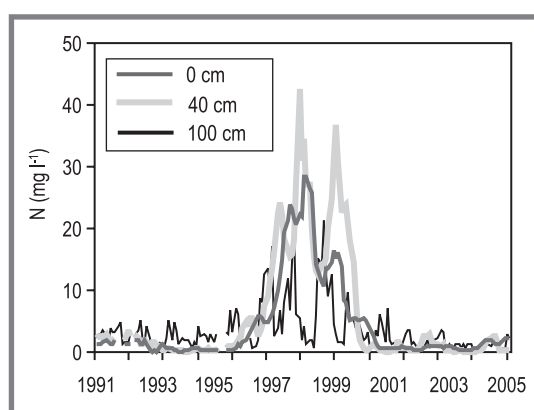


Figure 6.3.16 N concentrations (NO<sub>3</sub><sup>-</sup>, NH<sub>4</sub><sup>+</sup>) in seepage water at 0 cm, 40 cm and 100 cm depth on the spruce plot FI.

### Drastic changes in the balance of nitrogen and of nutritional base cations

Between 1992 and 1996, there was an output of  $5 \pm 2$  kg N ha<sup>-1</sup> a<sup>-1</sup> with seepage water, while net retention in this spruce ecosystem was 10 kg ha<sup>-1</sup> a<sup>-1</sup> (Figure 6.3.17). After the bark beetle attack in 1996, N losses increased rapidly to more than 200 kg ha<sup>-1</sup> a<sup>-1</sup> in 1998 and then decreased until 2002 (9 kg ha<sup>-1</sup> a<sup>-1</sup>). Within this period of six years excess mineralization, N losses added up to more than 500 kg ha<sup>-1</sup> a<sup>-1</sup>. In contrast, the breakdown of canopy structures minimized interception gains of aerosols and thus the deposition of N to  $11 \pm 2$  kg N ha<sup>-1</sup> a<sup>-1</sup>, which equals the rate of wet deposition. These losses, exceeding the previous “normal” export, summed up to about 494 kg N ha<sup>-1</sup> (1997 – 2002), which amounts to 38 % of the N pool in stand biomass and to 4 % of the ecosystem’s total reserves. Assuming that excess mineralization is now completed and that soil organic matter was the only source of these losses, leads to the conclusion, that 95 % of the soil N at this site is of recalcitrant quality. But it is unclarified whether this assessment – that there is a huge stable humus fraction – can be validated under conditions of chronic excess N deposition and ongoing changes in climate.

This six-year-period of excess mineralization also saw a dramatic increase in losses of nutritional base cations. Mean output rates were 6.4 (K<sup>+</sup>), 13.6 (Ca<sup>2+</sup>), and 7.3 kg ha<sup>-1</sup> a<sup>-1</sup> (Mg<sup>2+</sup>), which is two times (Ca<sup>2+</sup>) and three times (K<sup>+</sup>, Mg<sup>2+</sup>) higher than it was before the dieback. In order to demonstrate the relevance of these quantities for the

nutrition of the following tree generation, the difference of nutrient losses after (1997 – 2002) and before (1992 – 1996) the dieback was related to plant available pools in this system, i.e. total contents in stand biomass and exchangeable contents in the fine earth fraction of the soil.

$K^+$  and  $Ca^{2+}$  losses were small (2 and 4 % respectively) while  $Mg^{2+}$  losses amounted to 12 % of the system's reserves, of which the soil comprises only one third (Figure 6.3.18). But these losses via seepage appear small in comparison to the nutrient export by a hypothetical harvest: harvesting of timber only would reduce the plant available pool by 16 % ( $K^+$ ), 22 % ( $Mg^{2+}$ ), and 21 % ( $Ca^{2+}$ ), and with timber over bark, 23 % ( $K^+$ ), 33 % ( $Mg^{2+}$ ), and 48 % ( $Ca^{2+}$ ) would be removed.

Thus, the seepage losses do probably not affect the nutrient supply of the regenerating stand. And, without removal by harvest, additional nutrients will be available from slowly decomposing stand biomass. In managed forests, at sites of similar or even poorer fertility, nutrient losses via seepage of similar order have to be kept in mind in addition to the removal by harvest, after large-scale storm damages for example.

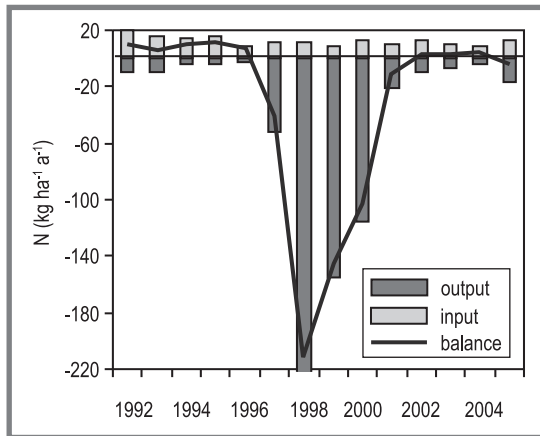


Figure 6.3.17 N mass balance for the spruce plot FI.

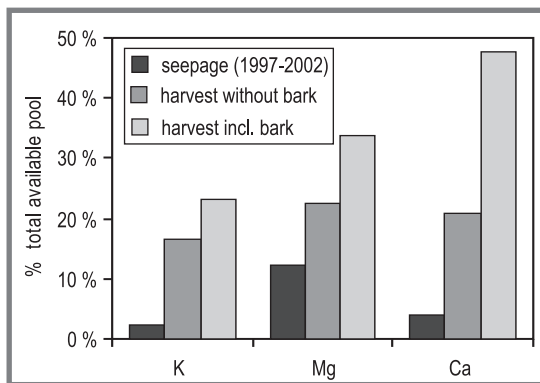


Figure 6.3.18 Bark beetle induced losses of base cations via seepage (% of available pool) in comparison with their export via hypothetical harvest at spruce plot FI.

### Hydrochemical and biological responses in groundwater and runoff

Effects and interactions of changes in deposition load and of altered biochemical processes, triggered by spruce dieback, have become apparent in groundwater and in runoff, concerning both, the cycling of water and solutes. Groundwater has been monitored in wells on beech plot B1 and in the adjacent Markungsgraben catchment at about 1000 m a.s.l. Discharge recording and sampling of water has been carried out at Schachtenau gauge (787m a.s.l.).

Between 1992 and 1998, mean annual runoff of Forellenbach (Figure 6.3.19) amounted to  $59 (\pm 3) \%$  of annual precipitation (1.56 m). Since 1999, when dead spruce stands accounted for more than 25 % of the area, runoff has increased to  $68 (\pm 4) \%$  of 1.66 m annual precipitation. This is due to reduced evapotranspiration losses from these sites, estimated at  $0.63 (\pm 0.07) \text{ m}$  (1992 – 1998) and subsequently at  $0.53 (\pm 0.13) \text{ m}$  by the difference depth. Additionally, maximum flood discharge and mean groundwater table increased slightly.

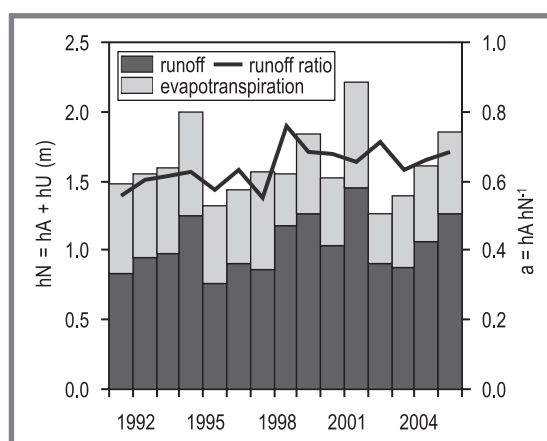


Figure 6.3.19 Water balance of the Forellenbach area (m), divided into runoff ( $h_N$ ) and evapotranspiration ( $h_U$ ) and runoff ratio (a).

### The increase of nitrate in groundwater and runoff influences sulphate dynamics

Up to 1997,  $\text{NO}_3^-$  concentrations in runoff water and in groundwater showed a slight decrease (Figure 6.3.20), at least partly the response to decreasing N deposition (s. a.). Parallel to the shift in runoff ratio,  $\text{NO}_3^-$  concentrations in runoff and in two out of three groundwater wells increased to  $7 - 8 \text{ mg l}^{-1}$ , indicating the import of highly concentrated soil water from disturbed spruce sites (s. a.). The lower-elevation groundwater catchment is apparently the most affected, since  $\text{NO}_3^-$  concentrations at this hillside have the highest median ( $20 \text{ mg l}^{-1}$ ) and the highest maximum value ( $28 \text{ mg l}^{-1}$ ).

Until 1998, sulphate concentrations in groundwater remained unchanged at about  $4 \text{ mg l}^{-1}$  and  $5 \text{ mg l}^{-1}$  respectively (Figure 6.3.21). The subsequent decrease to about  $3 \text{ mg l}^{-1}$  and  $4 \text{ mg l}^{-1}$  started in 1999, synchronically with the increase in  $\text{NO}_3^-$  concentrations. Concentrations in runoff already began to fall in 1996, reflecting the downward trend in soil water at undisturbed sites (s. a.). The mechanism behind these fast changes is the excess nitrification under dead spruce stands, which retarded the reduction of acidity, stored in Al and  $\text{SO}_4$  containing mineral phases in the soil during decades of much higher acid loads. But this retardation lasts presumably only as long as excess nitrification occurs on large areas of the catchment.

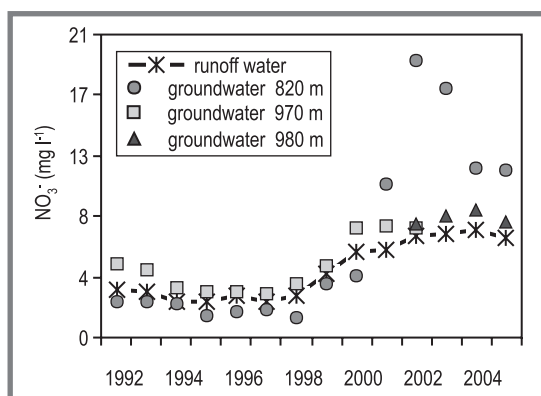


Figure 6.3.20  $\text{NO}_3^-$  concentrations in runoff water (discharge weighted means) and in groundwater (medians) at different elevations.

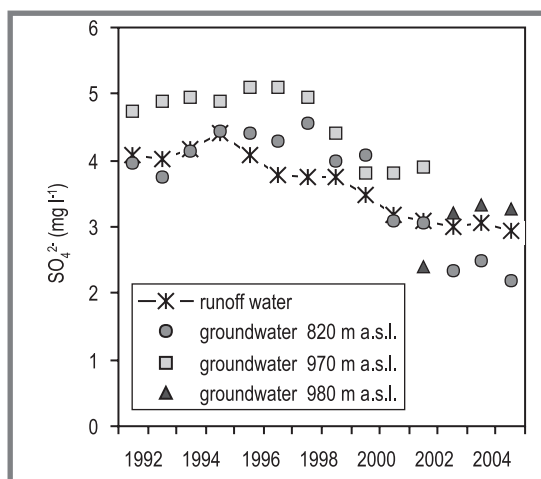


Figure 6.3.21  $\text{SO}_4^{2-}$  concentrations in runoff water (discharge weighted means) and in groundwater (medians). \* LFW (2004).

### Recovery from episodic acidification and its biological significance

The most important improvement of runoff chemistry is the reduction of episodic acidification at high flood, especially during snow melt. In the beginning of the 1990s  $\text{SO}_4^{2-}$  concentrations increased by about  $2.5 \text{ mg l}^{-1}$  as discharge increased by one order of magnitude (Figure 6.3.22). Since 2000 the slope of the concentration – discharge relation has fallen to below  $1 \text{ mg l}^{-1}$ .

Until 1999,  $\text{NO}_3^-$  concentrations showed no relation to the discharge. In 2000 and 2001,  $\text{NO}_3^-$  concentrations increased significantly with increasing discharge, indicating the input of highly concentrated soil water into the brook. Since 2002,  $\text{NO}_3^-$  concentrations have decreased significantly with increasing discharge; at present,  $\text{NO}_3^-$  concentration in groundwater determines the  $\text{NO}_3^-$  concentration in runoff.

$\text{Al}^{n+}$  and free acidity are significantly positively correlated with  $\text{SO}_4^{2-}$  (in 2000 and 2001 with  $\text{NO}_3^-$  too). It is well known, that low pH values and high Al concentrations severely affect the reproduction success of brown trout. The analysis of fish populations and water chemistry revealed a negative correlation between the maximum Al concentration during the period from 10 January and 9 February and the number of juveniles from the preceding spawning period (Figure 6.3.23). In response to hydrochemical recovery, the size of the brown trout population has increased markedly since 2000 (Figure 6.3.24). In addition, fish biomass has become at least two times higher than before. The biogeochemical effects of spruce dieback seem to interact positively with recovery from airborne acidification: input of litter, increased radiation and water temperature, increased  $\text{NO}_3^-$  and (to a smaller extent) base cations concentrations have all helped to significantly increase food supply to the fish.

Figure 6.3.22 The slope of annual concentration ( $\text{mg l}^{-1}$ ) – log discharge ( $\text{l s}^{-1}$ ) relations of  $\text{SO}_4^{2-}$  and  $\text{NO}_3^-$  in Forellenbach water.

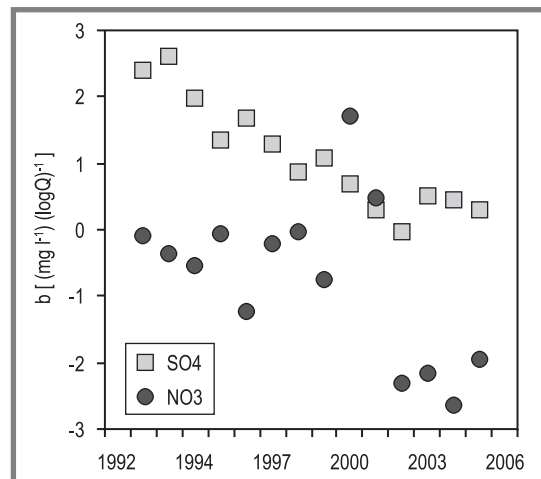


Figure 6.3.23 Number of juvenile brown trout (June –August) and maximum [Al] in runoff water from 10 January to 9 February.

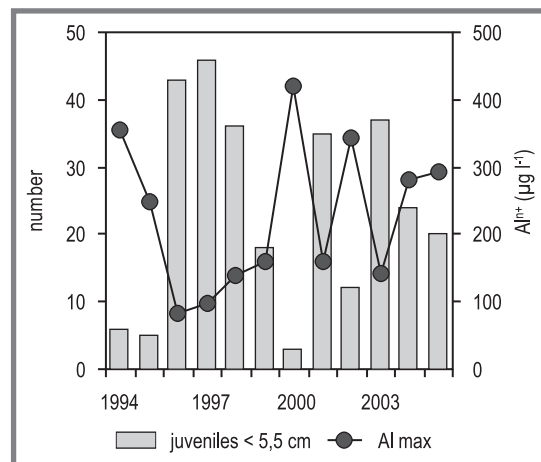
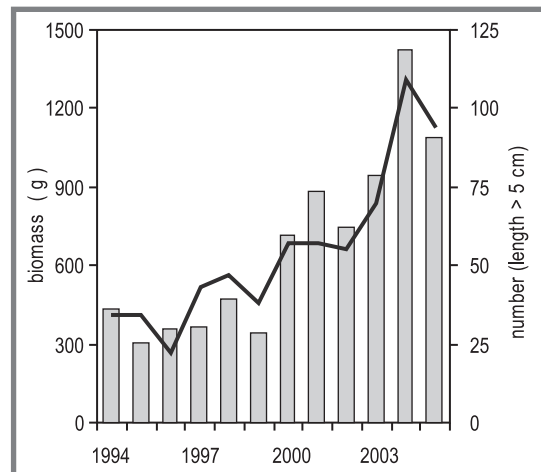


Figure 6.3.24 Size (right ordinate) and biomass (g fresh weight) of the brown trout population in the Forellenbach.



### 6.3.5

#### Modelling the water balance of monitoring plots and the Forellenbach area

The hydrological balance of the Forellenbach area has been modelled with the ArcEgmo-PSCN package (Klöcking *et al.* 2004, Klöcking 2006). It is a GIS-based, multi-scale modelling system for the spatially distributed simulation of hydrological processes in river catchments. In addition to usual model approaches for lateral surface and subsurface water flows at river basin scales (Becker *et al.* 2002, Pfützner 2003), it contains complex growth models for forests stands and crops and a detailed soil model (water, heat) to calculate the vertical water flows.



Based on GIS-maps and inventories, the study area has been divided horizontally into hydrotopes which are quasi-homogenous with regard to system properties (soil, stands, ground water table, gradients, and others). Air temperature, precipitation, humidity, and global radiation in daily resolution are the required climatic driving variables. Due to that architecture, plot measurement data can be used to validate the sub-models in addition to discharge comparisons.

For the beech stand B1 the simulated growth was successfully tested against measured parameters (Figure 6.3.25). The simulated water contents in different soil layers fit sufficiently the natural scatter of particular water contents measured by means of Time Domain Reflectometry. This was also true in the late summer 2003, when plant available water was exhausted (Figure 6.3.26).

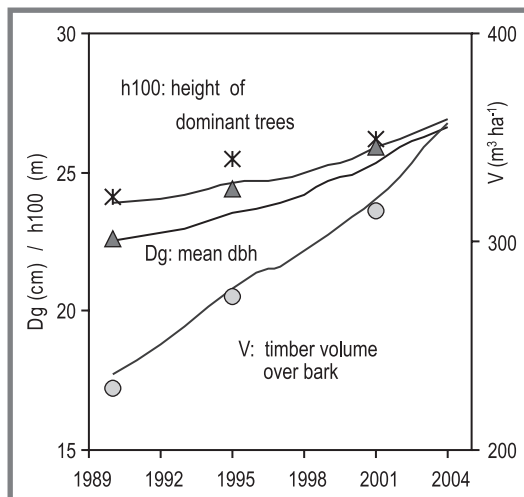


Figure 6.3.25 Comparison of simulated (lines) against measured growth parameters of the beech stand B1.

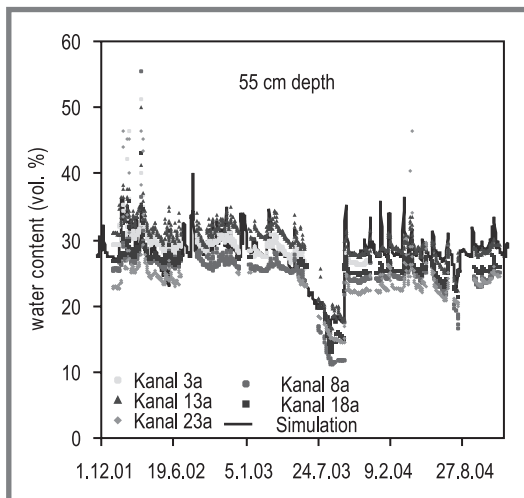


Figure 6.3.26 Comparison of simulated (line) against measured water contents at 55 cm depth (5 places in the beech stand B1).

The simulated water fluxes below the rooting zone (145 cm) were compared to groundwater table variations, which were recorded in the nearby well on the same plot. The course of modelled deep seepage (groundwater recharge) closely follows the course of the water table (Figure 6.3.27), indicating a realistic description of the ecosystem's vertical hydrological processes in this model.

A final step of model validation was the examination of mass balances, in particular that of chloride. Ulrich's approach was used to estimate total deposition from measured bulk and stand deposition. Seepage losses were calculated on annual (water flux times median concentration) and on monthly base (water fluxes times mean concentrations). The mass balances of all compounds do not differ between the procedures. The balances of chloride, which is considered to behave inertly in such ecosystems, are

close to zero (Figure 6.3.28). The mass balance approach was then applied to the spruce stand F1, to check the model performance on disturbed systems (Figure 6.3.29). Compared to the use of fixed rates of  $650 \text{ mm a}^{-1}$  (1992 – 1996) and of  $350 \text{ mm a}^{-1}$  (since 1997) chloride balances calculated with modelled evapotranspiration rates show a slight net output (1992 – 1996, 2001 – 2004). Coincident high net losses (1997 – 2000) indicate the release of organically bound chloride during excess mineralization. In conclusion, chloride balances have proved the model's ability to realistically reflect the water cycle in disturbed (F1) and undisturbed ecosystems (B1).

In addition to digital maps of topography, soil types, and related soil physics, the hydrological simulation of the catchment, which combines disturbed, undisturbed and mixed ecosystems, requires spatially discrete information about the vegetation cover. The species composition and biomass stocks of forest stands from inventories

Figure 6.3.27 Comparison of simulated groundwater recharge and measured groundwater table variations in the beech stand B1.

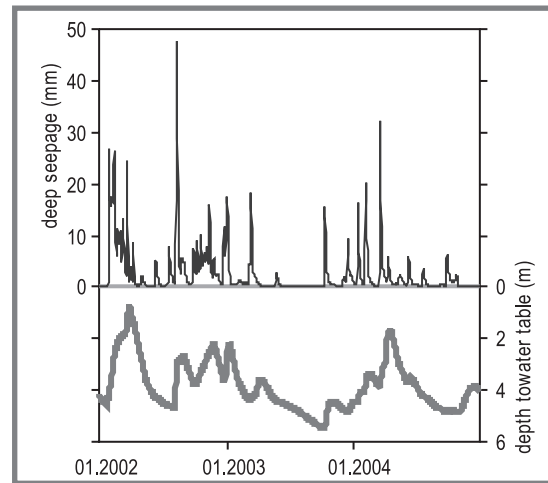


Figure 6.3.28 Mean annual input-output mass balance for the beech stand B1 (1992-2004).

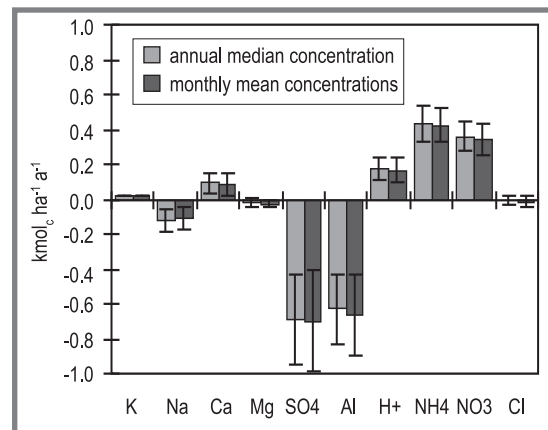
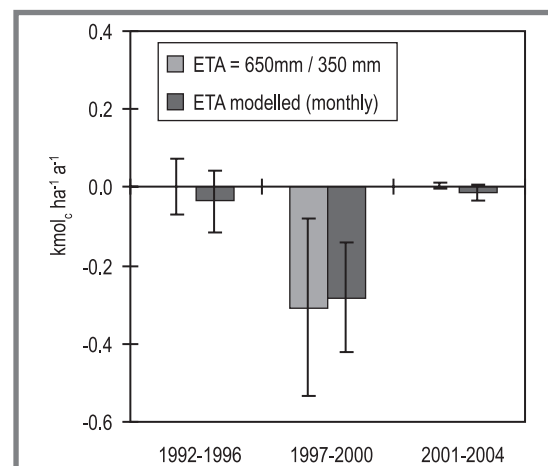


Figure 6.3.29 Mean annual chloride balance for the spruce stand F1 (1992-2004).



(1990 and 2001) were attributed to hydrotopes. But its changes in-between, caused by windthrow and bark beetle attack, were taken from annual aerial pictures, which by this means influence the water cycle on every particular hydrotope and the catchment (Figure 6.3.30). Modelling exercises with (reference) and without spruce dieback (scenario) gave insight into the influence of bark beetle's action on the water cycle of the Forellenbach area.

Simulated daily discharges are generally in good accordance with measured discharges, as demonstrated with scatter plot and duration curve (Figure 6.3.31). In addition, simulated monthly runoff fits very well with measured runoff ( $R^2 = 0.90$ ). But there's some inherent tendency to underestimate evapotranspiration and so to overestimate discharges, in summer mainly. In consequence, simulated mean annual output is about 12 % higher than mean measured output (1992 – 2004).

The analysis of runoff generation in the catchment area revealed, that surface (direct) and subsurface (hypodermic) runoff are of minor importance, contributing 2 % and 25 % to the total runoff respectively. Groundwater discharge was estimated to provide 73 % of total runoff. This result has been supported by procedures of chemical hydrograph separation, which showed groundwater discharges of about 80 % of total runoff. Despite of the important changes in canopy cover and water budget, the quantitative contribution of these discharge components has not significantly changed during the last 13 years.

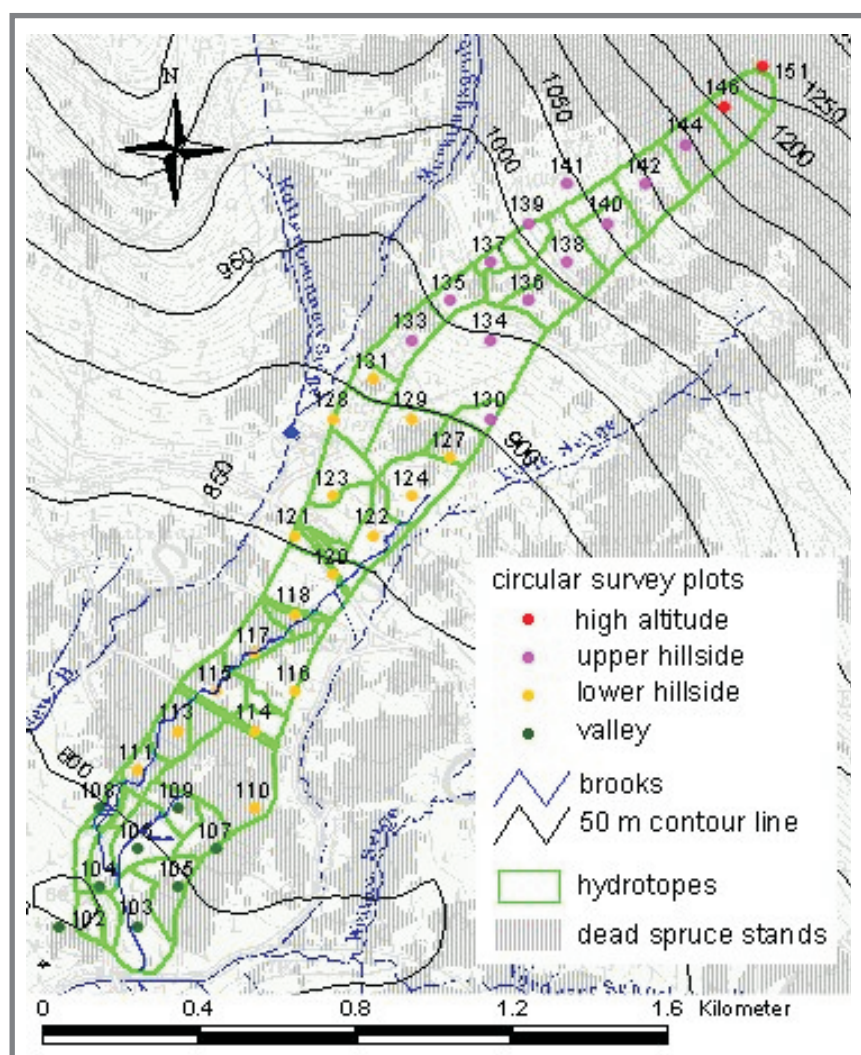


Figure 6.3.30 Topographic map of the Forellenbach area comprising inventory plots, hydrotopes, and dead spruce stands (hatched) from colour-infrared pictures (status 2004).

Up to 1998 small extents of dead spruce stands caused small reductions in interception and transpiration (Figure 6.3.32). As the share of dead spruce stands has reached about 20 % of total area (1999), interception and transpiration losses decreased by 60 – 80 mm a<sup>-1</sup>, while evaporation from soils, snow, and waterlogged depressions increased, as did groundwater recharge and runoff (~60 mm a<sup>-1</sup>) resp. In addition, increasing precipitation amounts and insolation in dead spruce stands led to a 9-days advancement of snowmelt induced highflood.

Since 2001 the water cycle seems to have stabilized at a new level. But young spruce and mixed stands, simultaneously growing on more than 40 % of the catchment area, will increase evapotranspiration losses. Therefore groundwater recharge and runoff will decrease to quantities smaller than under the cover of largely (very) old stands, irrespective of the predicted regional warming in the global climate change.

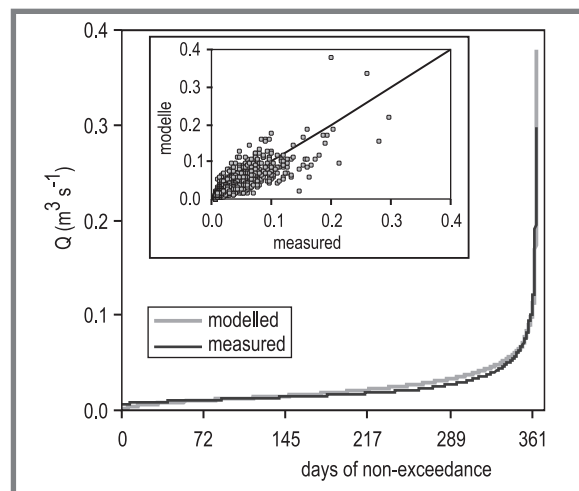


Figure 6.3.31 Scatter plot and duration curve of simulated versus measured daily discharges.

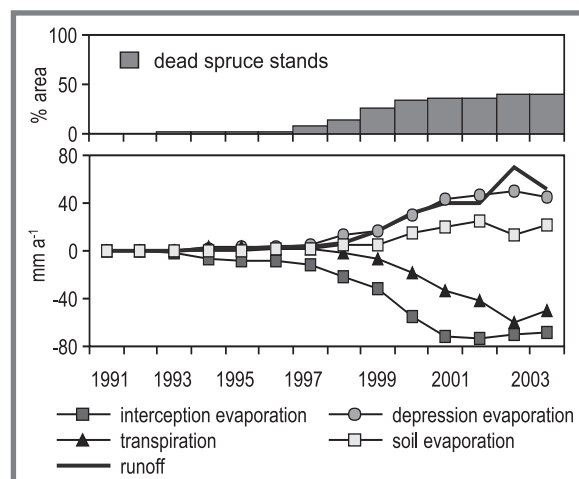


Figure 6.3.32 Dead spruce stands (% area) and bark beetle induced changes in the water cycle (mm a<sup>-1</sup>) of the Forellenbach area.

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6.4

## Report of national ICP IM activities in Latvia

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### Major activities under the programme in 2006-2007

The ICP IM programme work continued in 2006 through 2007 at the ICP IM sites of Rucava (LV01) and Zoseni (LV02), under the AM, AC, PC, TF, SF, SW, GW, RW, RB, LF, FC, AL, VG, FD, EP sub-programmes.

Meteorological measurements at the stations Rucava and Zoseni have been automated. Wind speed and direction, air and soil temperature, humidity, pressure, sunshine duration, total radiation and radiation intensity sensors were installed at the GAW/EMEP regional level station in Zoseni. Sensors of the same kind will be installed at Rucava in 2007. Automatic sensors for ultraviolet radiation and precipitation amount were acquired for both stations.

### Major programme activities focused at:

#### I. QA/QC of sampling and analysis

6 intercomparison exercises: 32<sup>nd</sup>, 33<sup>rd</sup> and 34<sup>th</sup> WMO/GAW Acid Rain, 24<sup>th</sup> EMEP NILU, 0620 ICP Waters (NIVA) and 8<sup>th</sup> needle/leaf interlaboratory comparison test. Of 261 analyses made, 42 results (16%) didn't meet the data quality objectives. Unstable were mostly measurements of low concentrations of Ca, Mg, Na, Ni, As, Pb, Zn



in precipitation and water. In 2007 the use of the inductive couple plasma mass spectrometer for heavy metals and the ion chromatography for base cations will greatly improve the determination results.

## II. Co-operative activities under the ICP Integrated Monitoring, ICP Forest, and ICP Vegetation programmes

Precipitation quality measurements have been made with similar sampling equipment and methods both at the ICP Forest II level and ICP IM sites. Comparison of deposition of major elements has shown lower deposition of **sulphur and nitrogen compounds** at the ICP Forest II level site (Central part of Latvia). The low deposition is caused by low precipitation that was 15% lower on average compared to the IM stations (Figure 6.4.1).

Observations of visible ozone injury to white clover under the ICP Vegetation programme were performed in 2006, 29 June through 5 October, at 5 sites, Zoseni, Rujiena, Dobeles, Mersrags and Rucava (Figure 6.4.2). Moss monitoring under the programme was performed in 2005/2006 on 101 polygons to cover heavy metals and nitrogen.

The white clover monitoring results from the 5 sites showed low concentrations of ground ozone. The degree of leaf damage exceeded 3 according to a scale of 1 to 6, and the ratio of ozone-sensitive clover to ozone-resistant clover biomass is as low as 0.6.

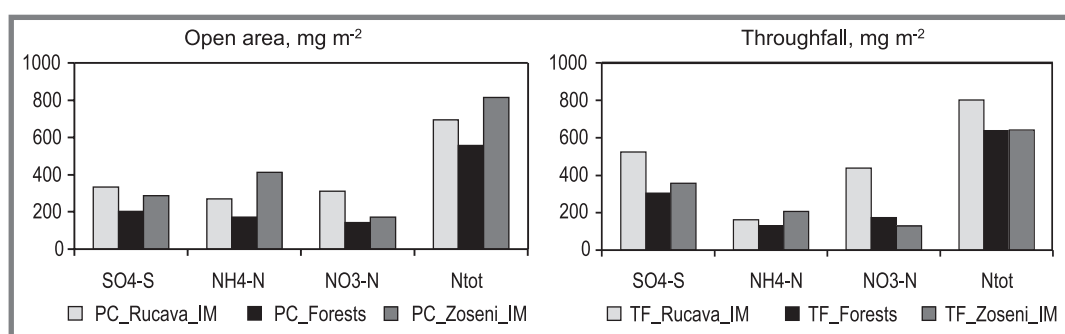


Figure 6.4.1 Wet deposition at the ICP IM and ICP Forest sites in 2005.



Figure 6.4.2 Ozone injury to white clover, experimental plot at the regional GAW/EMEP station Rucava.



Ozone injury to clover was the heaviest in the western part including Rucava as opposed to a less damaged continental part of the country, including Zoseni. (Nikodemus et al. 2006)

In 2006, air sampling with diffusive samplers was performed at 5 sites, GAW/EMEP regional level stations Rucava and Zoseni included, under a long-term research project "Evaluation of the temporal and spatial trends in the POPs ambient air concentrations in the countries of the Central and Eastern European region" aimed at the determination of effectiveness of the measures of international POPs conventions, organized by the POPs Centre, Brno, Czech Republic (Figure 6.4.3). Analysis of the measurement results will allow to identify the issues concerned with POPs and future monitoring strategy in this area in the country.



Figure 6.4.3 GAW/EMEP regional level station Rucava, POPs diffusive samplers.

### III. Some results of the interpretation of long-term biological and chemical data

Heavy metals in mosses, needles and litterfall and the tendencies in 1990-2006 have been analyzed.

Over the recent 10 years, since the establishment of integrated monitoring sites, heavy metal concentration in needles has not shown statistically great tendency to change. Yet, the concentration of Zn has increased slightly compared to the starting period of monitoring. This is a likely result of the increasing environmental effect of the Liepaja Metal Plant (Figure 6.4.4). (Nikodemus et al. 2006)

The concentration of Pb in litterfall has always been higher than in needles. Lead normally accumulates in the crown of a tree and reaches the upper layer of soil mostly with litterfall.

Due to transboundary pollution transfer and the effect of the Liepaja town's industries, Rucava has reported higher amounts of heavy metals (Pb, Zn, Cu, Cd) in litterfall from pine trees.

An analysis of the concentrations of Zn in litterfall at the station of Rucava for the recent 7-years period has shown a decrease in the concentration compared to 1999 and 2000 (Figure 6.4.5) (Nikodemus et al. 2006)

Mapping of mosses in the territory of Latvia has revealed zones of local pollution in the vicinities of Liepaja (Zn, Pb, Fe, Hg, As, Cd, Cr, N), Riga (N, Cr, Hg), Mazeikiai (V, Ni) and Naujoji Akmenes (N, Hg, Cu, Fe, Ni).

The heaviest pollution characterizes the vicinity of Liepaja mostly due to emissions from Liepaja Metal Plant and other industries. Of certain significance is the transboundary pollutant transfer. (Nikodemus et al. 2006)

During the period 1990-2005 there was a rapid decrease in the concentrations of Pb, Cr, Ni and V in *Pleurozium screberi* over the whole of Latvia while the concentration of Zn has increased during the last ten years in *Pleurozium screberi*. This trend is related to a rise in the production of the Liepaja Metal Plant (Figure 6.4.5).

A study of the changes in vegetation and environmental factors at the IM sites has shown regional differences in vegetation dynamics (Laivinš et al. 2007):

- eutrophication has been more intensive in Rucava than in Zoseni. It is obvious from both vegetation ordination data and the dynamics of Ellenberg indicator values and atmospheric depositions. In Latvia, this process has been more intensive in uplands (Vidzeme, Rietumkursā, Alūksne etc.), where spruce forest is at the final stage of oligomesotrophic pine forest succession. In the Coastal Lowland, invasion of spruce in pine forests is much slower. Spruce is more common in the field and shrub layers with some individual trees in the tree layer.
- slight changes in the Ellenberg's indicators (light, temperature, continentality, moisture, reaction, nitrogen). There was no significant temporal trend in temperature. The only significant trend was identified for moisture (positive) in Rucava (Peši plot) and for continentality (positive) in Zoseni. Rucava (Brušviti plot) has been more dynamic as five of the six Ellenberg's indicators have a positive or negative significant trend.
- gradual dynamics of sulphur and nitrogen in deposits. Such trends are observed also at other IM stations in Europe. On the other hand, trend in cations is positive both at Rucava and Zoseni. The positive trend is likely to retain in future as well due to intensive urbanization. It should be emphasized, that the amounts of sulphur, nitrogen and cations are higher in Rucava while the amounts of precipitation are very similar at both stations. (Laivinš et al. 2007)

Figure 6.4.4 The concentration (mg/kg) of Zn in the past year (C) and the previous year (C+1) in pine needles at Rucava (R) and Zoseni (T) IM stations, 1994 – 2006.

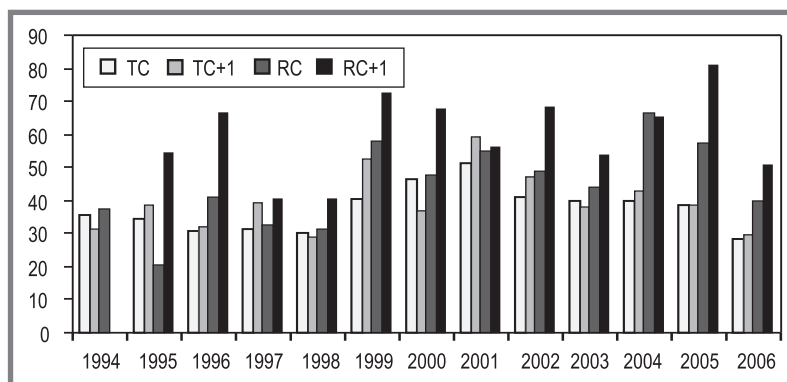
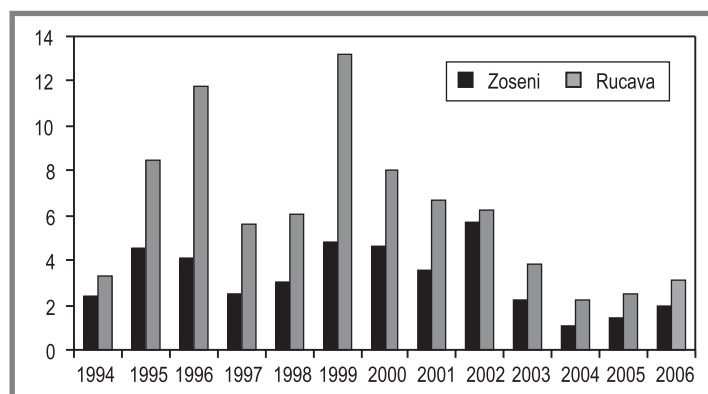


Figure 6.4.5 The concentration of Pb (mg/kg) in litterfall at Rucava and Zoseni IM sites.



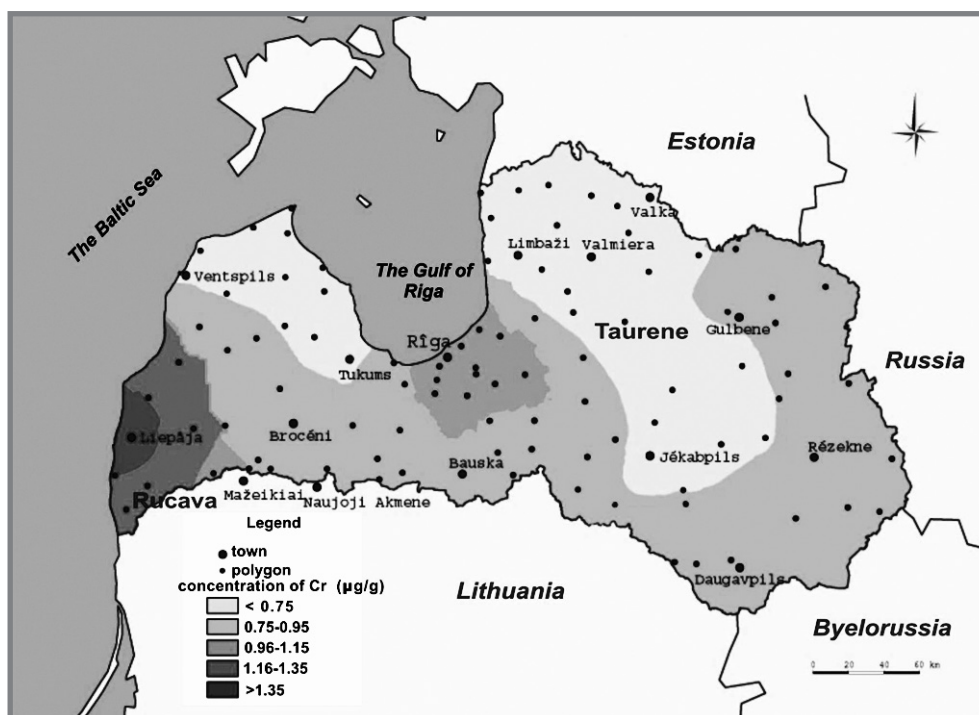


Figure 6.4.6 The concentration of Cr ( $\mu\text{g mg}^{-1}$ ) in mosses *Pleurozium schreberi* in Latvia 2005.

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## Future work

- 2006 ICP IM data reporting to the IM database;
- replacement of precipitation sampling equipment at both IM sites (open area, throughfall, stemflow);
- installation of an information board near the stations and publishing of a leaflet informing of the tasks and goal of the ICP IM sites (together with the State Forest Service of Latvia);
- co-operative analysis of precipitation and soil water quality measurement results under the ICP IM and ICP Forests programmes over the entire period of the observations;
- analysis of ground water quality at the IM sites based on the ground water quality data from the Latvia's hydrogeological network.

## Contact information

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#### Responsibility for the implementation of the ICP IM subprogrammes:

- Latvian Environment, Geology and Meteorology Agency: Climate, Air Chemistry, Precipitation Chemistry, Throughfall, Stemflow, Runoff Water Chemistry, Groundwater Chemistry, Hydrobiology of Streams.
- University of Latvia (Dr. O. Nicodemus): Soil, Soil Water, Litterfall Chemistry, Foliage Chemistry, Metal Chemistry of Mosses.
- University of Latvia (Dr. M. Laivinsh): Vegetation, Forest Damage, Trunk Epiphytes, Forest Stand Inventory, Vegetation Structure and Species Cover.
- Responsibility for the implementation of the ICP Forests programme at Level II: State Forest Service (Ieva Zadeika)
- Responsibility for the implementation of the ICP Modelling and Mapping programme: Latvian Environment, Geology and Meteorology Agency (Iveta Shteinberga)

6.5

### Report on national ICP IM activities in Lithuania

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#### Introduction

Sulphur emissions have declined in Europe by 67% since the early 1980s due to enactment of strict pollution control strategies whereas nitrogen emissions remained fairly constant. Therefore, even after a complete implementation of the Gothenburg Protocol and other current legislation the effect of N deposition with commensurate adverse biological effects still remains the most relevant problem to address in Europe as well as in the USA and Canada.

The study focuses on the analysis of specific effects of  $\text{NH}_4^+$  and  $\text{NO}_3^-$  concentration in the air and their deposition on N enrichment processes in N limited forest ecosystems.

#### N concentration in the air and N atmospheric deposition fluxes

The most significant decrease in nitrogen compounds in the air lasted until 2001.  $\Sigma\text{NH}_4^+$  concentration in the air at LT03 decreased by 86%, at LT02 by 65%. In 2002, a slight increase in the concentration was recorded. Afterwards,  $\Sigma\text{NH}_4^+$  concentration in the air was stable at the level of 1.1 – 1.3  $\mu\text{g N m}^{-3}$  both at LT01 and LT03.

Although annual means of  $\Sigma\text{NO}_3^-$  concentration in the air were quite stable 0.5-0.7  $\mu\text{g N m}^{-3}$  over the entire period considered at all stations, an increase in mean annual concentration of  $\Sigma\text{NO}_3^-$  since 2001 has been observed (Figure 6.5.1).

The changes in annual wet deposition for the period 1994-2005 had a very similar pattern to that in the air: a decrease in annual wet deposition of  $\text{NH}_4^+$  and no significant change in wet deposition of  $\text{NO}_3^-$  (Figure 6.5.2).

Analysis of the spatial pattern of regional pollution level revealed that western and southern parts of Lithuania were more polluted by N compounds, which is most likely related to the proximity of these areas to the major pollutant sources in Central Europe as well as to the difference in the amount of precipitation.

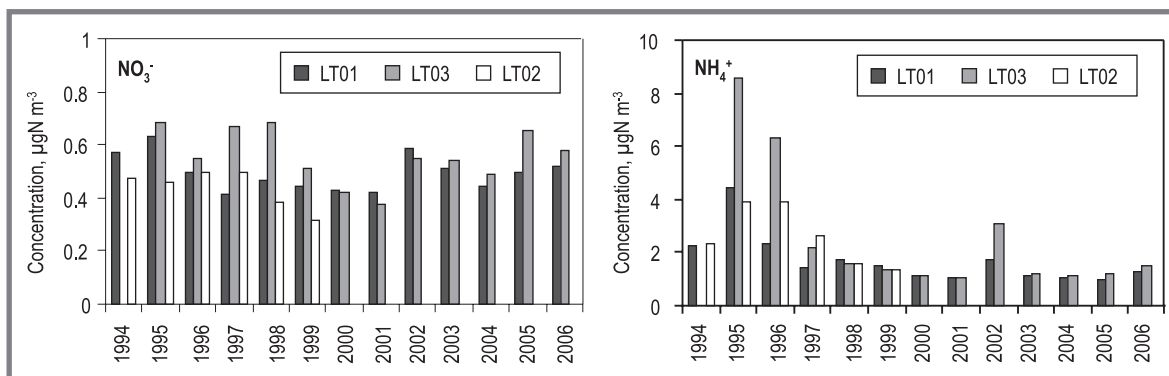


Figure 6.5.1 Changes in mean annual concentrations of  $\Sigma\text{NH}_4^+$  and  $\Sigma\text{NO}_3^-$  in the air 1994-2005. ( LT01 – Aukstaitija IM site, LT02 – Dzukija IM site, LT03 – Zemaitija IM site)

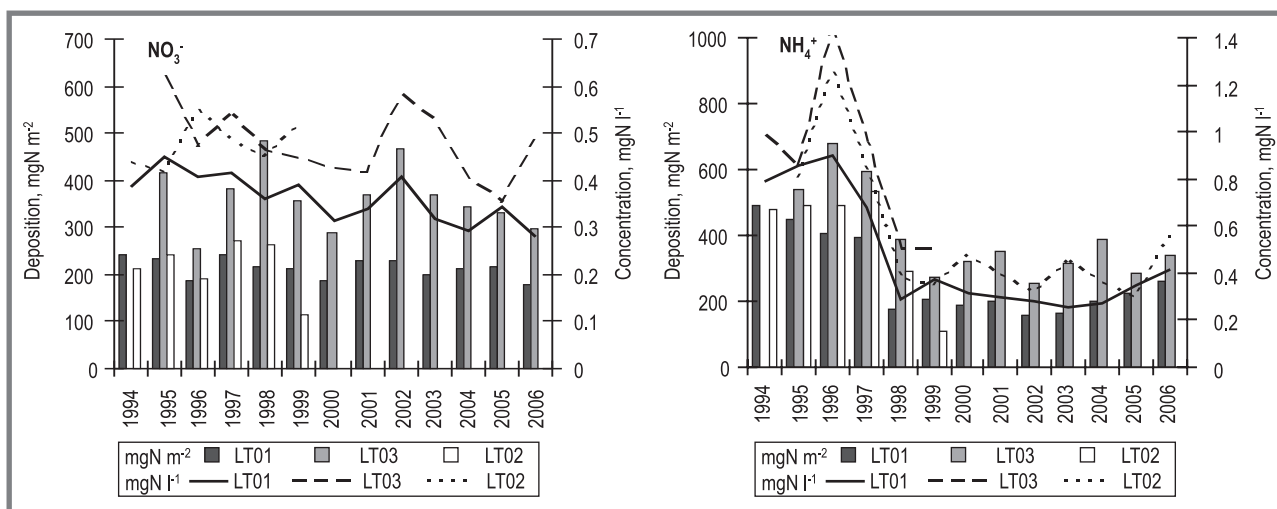


Figure 6.5.2 Changes in  $\text{NH}_4^+$  and  $\text{NO}_3^-$  depositions, and concentrations in precipitation. ( LT01 – Aukstaitija IM site, LT02 – Dzukija IM site, LT03 – Zemaitija IM site)

### N concentrations in soil, ground and runoff water and their flows

In the first part of the period considered, i.e. from 1994 to 1999, soil water at LT02 was more contaminated with  $\text{NO}_3^-$  than in the other stations and demonstrated a significant upward trend. At LT01 the upward trend for the period 1994-2003 was not as significant. Since 2004, however, a considerable increase in concentrations has been observed. At LT03 a downward trend in  $\text{NO}_3^-$  concentration in soil water was detected (Figure 6.5.3).

In LT01 a significant downward trend in  $\text{NH}_4^+$  concentration in the soil water was detected. Similar changes in this compound were detected at LT02 for the period, when it was in operation. Contrary to these, at LT03 no significant trend was detected (Figure 6.5.3).

In the first years of observation (1995-1997) flow of  $\text{NO}_3^-$  in the soil water of all IM stations were the lowest: 5-10 mg N  $\text{m}^{-2}$  per year. Afterwards, at LT01  $\text{NO}_3^-$  flow in soil water had a tendency to increase and over the last 2-year period 2004-2005 reached the highest level, 80-120 mg N  $\text{m}^{-2}$  per year, i.e. increased 8-12 fold if compared to 1995-1997. Changes at LT03 were not so pronounced, however, started earlier and lasted longer – in 1997  $\text{NO}_3^-$  flow increased up to 22 mg N  $\text{m}^{-2}$  per year. In 2001  $\text{NO}_3^-$  flow reached the maximum value over the entire observation period 24-47 mg N  $\text{m}^{-2}$ , i.e. 2-4 fold exceeded the flows of 1995-1996. Contrary to LT01, in 2002-2005 the flows of  $\text{NO}_3^-$  in soil water at LT03 decreased.

N concentrations in ground water were investigated at 4 bores in each stations.  $\text{NO}_3^-$  concentrations in the ground water of LT01 had no statistically significant trends, with the exception of bore No 2, where over the period from 1999 to 2002 its concentration increased drastically (Figure 6.5.4). These changes resulted in the highest  $\text{NO}_3^-$  flow which lasted till 2005. In the rest of bores, the changes were insignificant. At LT03  $\text{NO}_3^-$  concentrations in ground water of the shallow bores had a tendency to decrease, whereas in the deeper bores – a tendency to increase (Figure 6.5.5). These changes resulted in the highest  $\text{NO}_3^-$  flow in the deeper bores in 1998, 2002 and 2005, and in the shallow bores in 1996-1997.

$\text{NH}_4^+$  concentration in ground water had a tendency to decrease in all bores of all IM stations, however,  $\text{NH}_4^+$  flow at LT01 had a tendency to increase, reaching the highest values in 1998, 2002 and 2004. Meanwhile at LT03  $\text{NH}_4^+$  concentration in ground water had a tendency to decrease (Figure 6.5.4 and 6.5.5).

The comparison of the means of concentrations of considered chemical components in soil and ground water of all three stations over the considered period, revealed higher concentrations of many parameters at Dzukija (LT02). Most likely it might be attributed to good filtrational features of the continental dune sand. Recently, higher concentrations of these components have also been observed at LT03.

Concentration of  $\text{NO}_3^-$  in runoff water had no significant trends over the considered period at all stations. However, some increase at LT01 and LT03 from 1994(95) to 1999(2001) and some decrease afterwards until 2005 was observed. Concentration of  $\text{NH}_4^+$  in runoff water had a tendency to decrease at all considered stations over the entire observation period (Figure 6.5.6).

Despite similar character of the changes in  $\text{NO}_3^-$  and  $\text{NH}_4^+$  concentrations in runoff water, there was an evident difference in their output. Over the period considered output of both N compounds had a tendency to decrease at LT01, but at LT03 to increase.

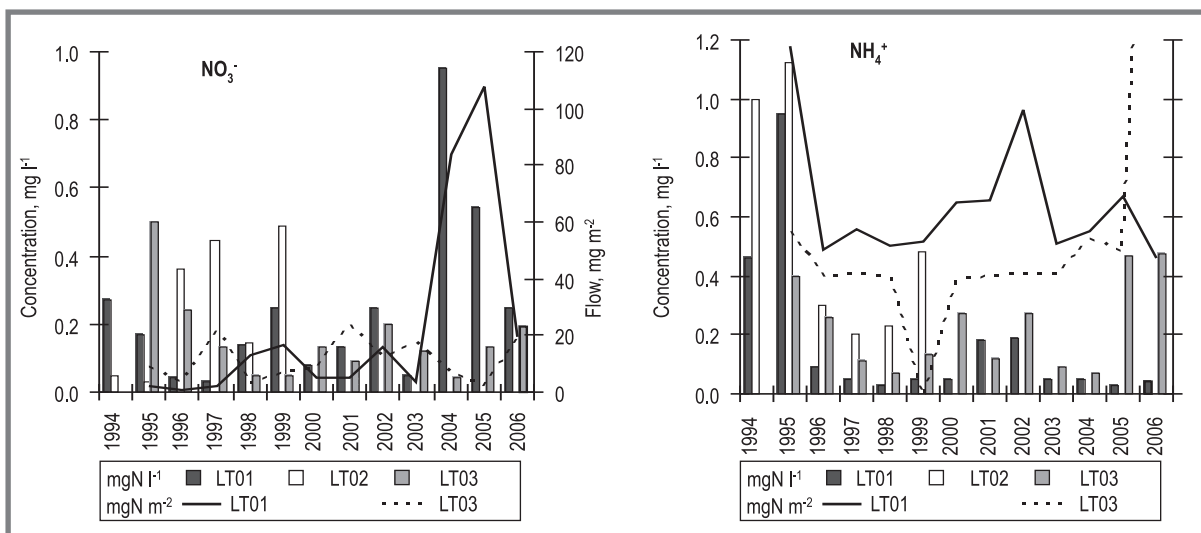


Figure 6.5.3 N concentrations in soil water and flows at 20 cm depth at IM sites.



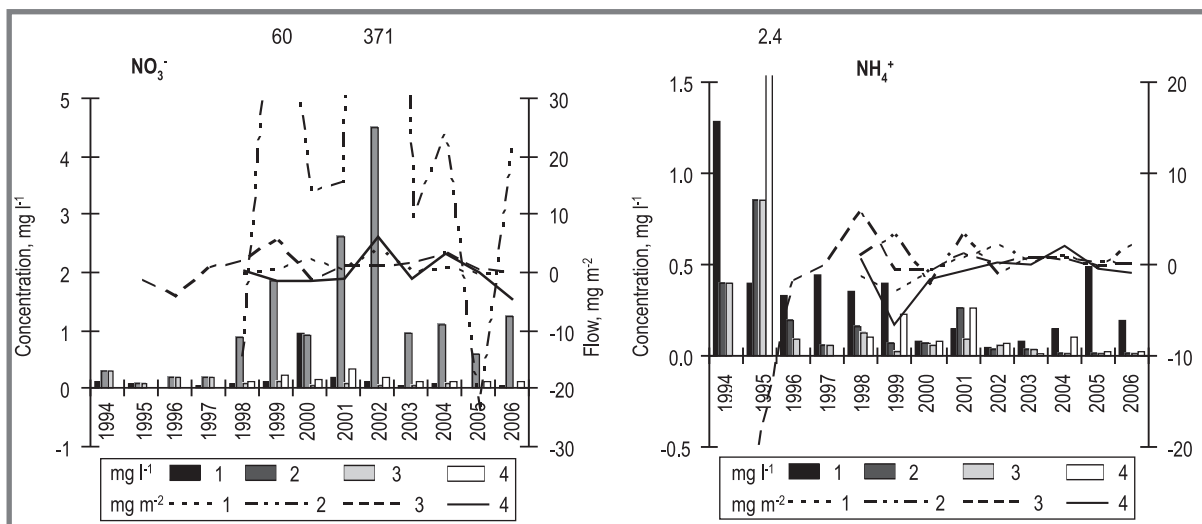


Figure 6.5.4 N concentrations and flows at different ground water depths at Aukstaitija IM site (LT01).

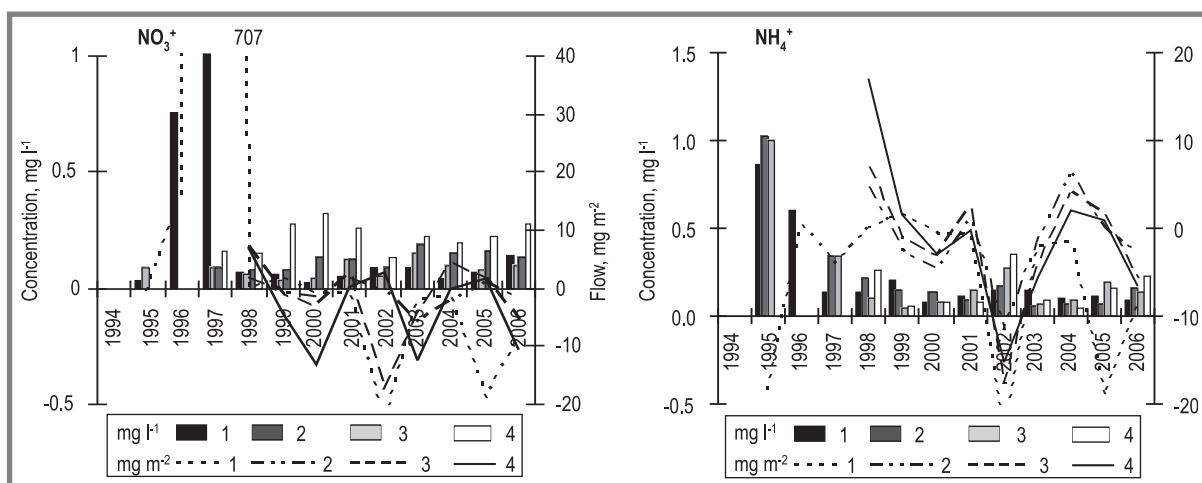


Figure 6.5.5 N concentrations and flows at different ground water depths at Zemaitija IM site (LT03).

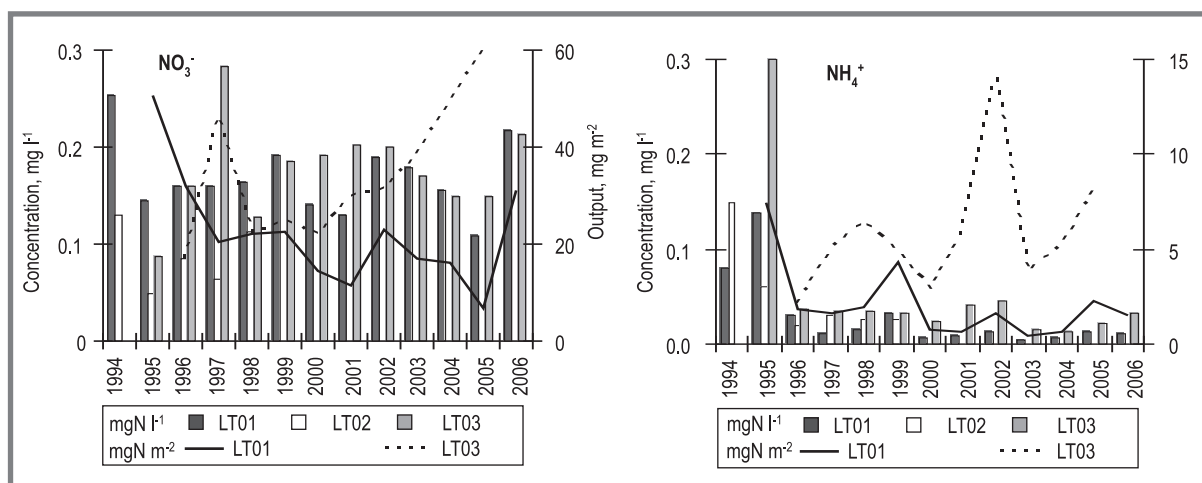


Figure 6.5.6 N concentrations in runoff water and their output at IM sites.

### **N mass balance**

The observed data revealed that nitrogen balance on all IM sites exhibited obviously similar trends – the amount of nitrogen inflow into the system was much greater than its elimination from the catchments. At LT01 the content of nitrogen output made up only 2-5% of the input, at LT03 2-4%. The variations between N input and output at the IM sites were rather small. The amount of precipitation had no influence. The great difference between the high input of nitrogen (in the form of compounds) and its low output from the system may be accounted for the fact that a great part of nitrogen is fixed in the biosphere.

The peculiarities of concentrations of nitrogen and sulphur compounds in precipitation may be explained by western transport which partly refutes the common assumption that the greatest amounts of acidifying chemical components in precipitation are transported to Lithuania with south-easterly winds. The outputs of nitrogen with runoff water on IM sites are rather variable. Individual features of the catchments – vegetation specifics were key contributing factors. At LT01 and LT03 the nitrogen balances reveal great amounts of nitrogen accumulated in the investigated geosystems – about 1000 kg km<sup>-2</sup> a<sup>-1</sup> at LT01 and about 600 kg km<sup>-2</sup> a<sup>-1</sup> at LT03.

### **6.6**

## **Report on national ICP IM relevant activities in Norway**

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### **Introduction**

The monitoring of air pollutants and their effects on ecosystems in Norway constitutes a comprehensive activity, with monitoring programmes on air quality, surface water, soils, forests and fauna (aquatic and terrestrial). Several institutions are involved to support the activities aimed to support the Convention on Long-range Transboundary Air Pollution (LRTAP Convention) and its Working Group on Effects (WGE). Studies of atmospheric deposition, surface water chemistry, aquatic biology and forest condition are performed at approximately 20 sites to support the ICP Waters and ICP Forest programmes respectively. From two of these sites (Birkenes and Kårvatn) data are also reported to support the ICP Integrated Monitoring. In general, all available data derived from these activities are used to evaluate cause-effect relationships, while specific evaluations based on ICP IM data alone have not been prioritised. In this note, a general description on the WGE related activities at Norwegian sites is presented.

### **Results from the monitoring effects of long range transboundary air pollutions in Norway 2005**

#### **Air and precipitation**

Emissions of SO<sub>2</sub> in Europe have decreased by about 66% since 1980, 56% since 1990. The emissions of nitrogen oxides and ammonia increased up to 1990 but have decreased since then by about 27 and 26% respectively (EMEP Status report 1/2005). The observed reductions in concentration levels are in agreement with the reported downwards trends in pollutant emissions in Europe. Since 1980 the content of sulphate

in precipitation decreased by 64-77%. Similar reductions in airborne concentrations were between 72- 92% and 65-73% for sulphur dioxide and sulphate, respectively. The nitrate and ammonium concentrations in precipitation have significantly decreased at most sites in southern Norway. There are, on the contrary, no observed significant trends for the nitrogen species in air, except for a clear decrease in the NO<sub>2</sub> concentration. ECE's critical level for accumulated ozone exposure above the threshold of 80 µg m<sup>-3</sup> (40 ppb) (termed AOT40) of 10.000 ppb hours for forests was not exceeded at any of the stations in 2005 nor threshold limit for accumulated ozone exposure of crops (3000 ppb hours). The highest annual mean concentrations of most of the heavy metals in precipitation were measured in Sør-Varanger (Svanvik) due to emissions in Russia. The heavy metal concentrations have generally decreased by about 60-80% from the late seventies, but after 1990 the concentration level has been relatively constant.

### **Water chemistry**

The decrease in sulphate in deposition has caused a decrease in sulphate in lakes and rivers of 34-74% from 1980 to 2005. 2005 in general shows the lowest sulphate concentrations in lakes and rivers measured during the monitoring programme (since 1980). As a consequence, the acidification situation in lakes and rivers has shown a clear improvement in the 1990s with increases in pH and ANC (Acid Neutralising Capacity) and a decrease in inorganic (toxic) aluminium. There is a marked shift in nitrate concentrations in the period 1997 to 2005, compared with the years before 1997. There is a general decrease in nitrate in most regions, although the yearly changes are small. The slight increase in TOC during the 90s has now levelled off, but the overall changes from 1990 to 2005 are still significantly increasing for most regions in Norway.

### **Fish**

The current status of fish populations in Norwegian lakes greater than 3.0 ha have been assessed in relation to effects of acidification during recent years. The number of lost and damaged populations of the six most common species of fish were estimated to be about 9.600 and 5.400, respectively. Brown trout has suffered the most severe damage with a total of about 8.200 lost stocks. Lakes in southernmost Norway (Agder) have suffered the highest damage with about 5.000 lost trout stocks. Test-fishing with gill nets in lakes throughout Norway, indicate an increase in fish abundance in most areas. However, some fish populations are still low in abundance, which can be due to acidification. The density of young brown trout in tributaries to lakes in Vikedal and Bjerkreim watersheds in south western Norway (Rogaland County) has increased significantly in recent years. Corresponding densities of young brown trout in Gaular watershed in western Norway have been more unstable, however, an increase in abundance was registered in 2005.

### **Forest**

Since 1997 the crown condition of trees has been largely stable. For Norway spruce and Scots pine the crown condition was slightly reduced in 2005, while it was slightly improved for birch. The reduction is most evident in the counties of Agder and Oppland in the south eastern part of Norway. Crown condition is determined by a number of factors and stresses, such as age, diseases (e.g. various fungi), growth conditions and climatic stress (drought and frost). When trees show signs of poor health, this is often due to an interaction of some of these natural causes. The variation we have seen in the last years has mainly been caused by fungi and insect attacks that were largely due to a combination of climatic stress to trees and a favourable climatic environment for the fungi and insects. Effects of air pollutants may come in addition

to or interaction with these factors. The effect of pollutants on forest condition is hard to estimate, because it has been small compared with those of other factors. Results from ecological investigations on the intensive monitoring plots suggest that the forest environment is stable, and that there are, as usual, large fluctuations from year to year in some measurements, within the normal variation for coniferous forests.

#### **Terrestrial flora and fauna**

As expected, documented changes in the ground vegetation of birch forests at the Børgefjell monitoring site did not show any effects related to long-range pollution but seem to be caused by local natural and anthropogenic disturbances to the vegetation. Inventories of epiphytic vegetation on trunks of birch at the monitoring sites (pine at Solhomfjell) show a clear relationship between lichen coverage and damage status and deposition patterns of pollutants, with the lowest coverage and highest damage frequency in the southernmost sites. Repeated inventories after 5, 10 and 15 years indicate generally improved coverage and damage status in the southern areas, as confirmed for the southern Solhomfjell site in 2005. A rather mild and moist climate during the monitoring period is the likely cause of the changes in the species assemblage of lichens on trees in Børgefjell.

Golden eagles and gyrfalcons at the monitoring sites exhibited similar patterns of production at southern polluted sites compared to northern sites. There is no indication that population variation in passerine birds is significantly different in southern compared to northern sites. Hatching success of pied flycatchers has been at comparable levels in southern and northern sites for several years.

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SFT 2006. Monitoring of Long Range Transported Air Pollutants and Precipitation. Annual report 2005. Ed Skjelkvåle, B.L NIVA. Norwegian Pollution Control Authority (SFT), Oslo. Report 957/2006. (in Norwegian)

#### **Other relevant reports, articles and conference abstracts using ICP IM data**

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- Kaste, Ø., Rankinen, K. and Lepistö, A. 2004. **Modelling impacts of climate and deposition changes on nitrogen fluxes in northern catchments of Norway and Finland.** *HESS*, 8: 778-792.
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## Report on national ICP IM activities in Sweden

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### Introduction

The Swedish integrated monitoring programme is run on four sites distributed from south central Sweden (SE14) over the middle part (SE15), to a northerly site (SE16) representing north Sweden. The long-term monitoring site SE04 Gårdsjön F1 is complementary on the inland of the West Coast and has been suffering from long-term high deposition loads. The Swedish group now compiled results from the four Swedish IM sites for the year 2005. The sites are well-defined catchments with mainly coniferous stands dominated by bilberry spruce forests on glacial till deposited above the highest coastline, meaning no water sorting of the soil material. Forest stands are mainly over 100 years and at least three of them have several hundred years of natural continuity and were up to c. 50 years ago partly lightly grazed woodlands. Both climate and deposition gradients coincide with site distribution from south towards north (Table 6.7.1).

Table 6.7.1 Geographic location and long-term climate at the Swedish IM sites.

	SE04	SE14	SE15	SE16
Latitude; Longitude	N 58° 03' ; E 12° 01'	N 57° 05' ; E 14° 32'	N 59° 45' ; E 14° 54'	N 63° 51' ; E 18° 06'
Altitude, m	114-140	210-240	312-415	410-545
Area, ha	3.7	19.6	19.1	45
Mean annual temp., °C	+ 6.7	+ 5.8	+ 4.2	+ 1.2
Mean annual precipitation, mm	1000	750	900	750
Mean annual evapot., mm	480	470	450	370
Mean annual runoff, mm	520	280	450	380

In the following, some special conditions and ongoing work for the four Swedish IM sites 2005 are presented.

### Climate and Hydrology

Climate for the year 2005 showed similar pattern as the previous year 2004. Temperatures were higher compared to long-term averages for the northern stations SE15 and SE16 with +0.4 °C and 1.0 °C, respectively. At the southern Sweden IM sites SE04 and SE14, the temperatures were lower compared to long-term averages with 0.4 °C at SE14. Temperature distributions for the year for the southern IM site showed colder summer but warmer winter. In central Sweden (SE15), autumn was warm as well as at the northern IM site SE16 where November and December had a temperature excess of 4 degrees.

Precipitation conditions could indicate the temperature with higher precipitation in summer for both southern and northern sites. However, periods with low precipitation

also occurred and altogether the values stayed rather normal with perhaps slightly higher precipitation at the northern site SE16. An exceptional rainfall event occurred in January on the south-western SE04 site with an excess of 85 mm, i.e. 156 mm 220% of normal.

The characteristic annual hydrological patterns of the catchments are high groundwater levels during winter and lower in summer and early autumn. This pattern should mainly also be reflected in runoff. However, warm periods in winter at the northern site SE16 furnished snowmelt and runoff in winter reflected in a fairly low spring snowmelt period. Runoff made up 36-61% of 2005 annual precipitation that could be compared to 2004 year values on 37-58% and the 2002 values on 40-47% and 2003 with 26-70%. High rates occur especially in the colder northern site and the south-western SE04 site (47%) while the other two Swedish sites showed similar values of 36-37%. Mainly, the runoff could be considered normal, only SE15 had on a slightly low runoff (250 mm) and the northern SE16 had high runoff (500 mm) (Table 6.7.2).

**Table 6.7.2 Compilation of the 2005 water balances for the four Swedish IM sites,**  
P – Precipitation, TF – Throughfall, I – Interception, R – Water runoff.

	Gårdsjön SE04		Aneboda SE14		Kindla SE15		Gammtratten SE16	
	mm	% of P	mm	% of P	mm	% of P	mm	% of P
Precipitation, P	1087	100	669	100	775	100	825	100
Throughfall, TF	742	68	444	66	476	61	569	69
Interception, I	345	32	225	34	299	39	256	31
Runoff, R	508	47	244	36	268	35	502	61
P-R	579	53	425	64	507	65	323	39

A special investigation on the importance of water pathways for element transport in catchments has been carried out in the Swedish IM site SE15 Kindla. This investigation was mainly directed on Al generation and transport. The outcome should consider a most appropriate location of the sampling places for soil, soil moisture and groundwater.

Transects from the very near-stream zone up to 50 m distance from the stream with drained soil types were instrumented with sampling devices and a number of water sampling occasions occurred. Results show important links between pH and Al occurrence but also strong influence from organic material (TOC) especially for the flow of organic and total Al (Figure 6.7.1).

Inorganic Al was generated in the drained upslope locations and transported to the near-stream zone where organic material in the soil increased. This organic material captured Al and incorporated it in organic bound form. With lateral water flow in the upper organic rich horizons, total Al could easily be transported to the stream. However, at rainfall events causing temporal acidic conditions, the inorganic Al could be released and added to the stream water. The storage would exert a potential source of Al. The temporal changes in hydrology would influence the flow and this could also concern other elements than Al. These results would increase the knowledge of cause effect relationships and provide better understanding of the effects from air pollution on water conditions.



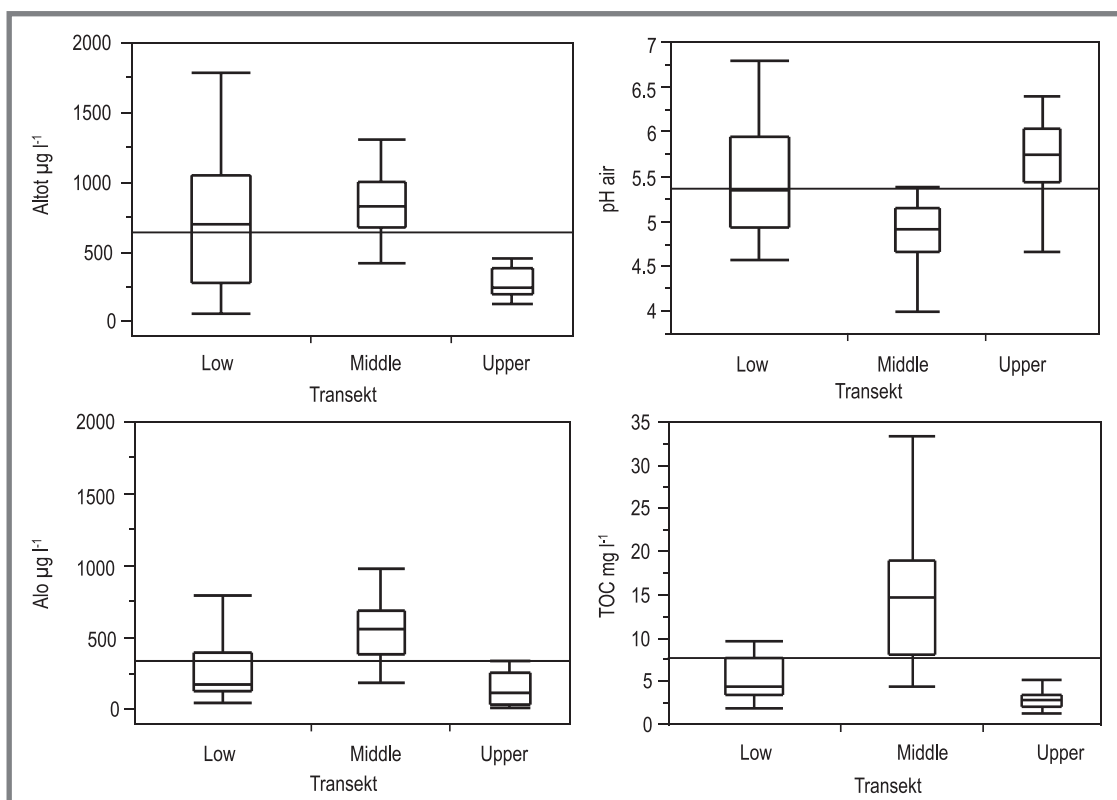


Figure 6.7.1 Distributions (Box plot) on concentrations of Altot and Ato ( $\mu\text{g l}^{-1}$ ) together with TOC ( $\text{mg l}^{-1}$ ) and pH in the three transects (Upper, Middle och Low) in Kindla in May and November for the years 2004-2006.

### Water chemistry

Low ion content characterises the deposition and throughfall for the three inland sites (c. 2 mS/m) while sea salt provides higher ion content in the west coast SE04 site (TF 6.7 mS/m). Water pathways through the soils of the catchments are fairly short and most of the surface water formation depends on short connections between infiltration and surface water formation. Acidity in the deposition was mainly the same at all sites with slightly higher (0.1 - 0.3 units) pH in throughfall compared to bulk deposition, i.e. pH c. 4.8 (Table 6.7.3). Chemical reactions during water flow through the catchments buffered the acid water fairly little and pH in the stream water were below 4.6 in the three southern sites while in the northern site SE16 pH in stream water was c. 5.7 with an ANC of c. 0.09 meq  $\text{l}^{-1}$ . At SE14, the pH did not change very much from deposition to stream water but ANC was added dependent on increased DOC and reached about 0.10 meq  $\text{l}^{-1}$ . Organic anions contribute to positive ANC in three of the catchment stream waters but in SE15 with low DOC, the ANC is negative (c. -0.01 meq  $\text{l}^{-1}$ ).

Anion deposition varied between the sites with Cl being the dominant ion except for the northern site SE16. In the soil water, the dominating inorganic anion was  $\text{SO}_4$  in the two northern sites. Chloride showed highest stream water values in SE04 and SE14. Sulphate was released from the soil and provided higher values as compared to deposition in stream water at all sites. Organic anions dominated stream water in SE16 while it only reached up to 25% of the anion content at the other sites.

Table 6.7.3 Atmospheric deposition chemistry for the four Swedish IM sites, S and N in kg ha<sup>-1</sup> a<sup>-1</sup>.

	SE04	SE14	SE15	SE16
pH, bulk deposition	4.8	4.8	4.8	4.9
pH, throughfall	4.7	5.1	4.9	5.0
SO <sub>4</sub> -S	15	12	8	5
N-tot	9.0	8.6	4.9	2.8

#### The storm 'Gudrun' resulted in windthrown trees

In early 2005, a heavy storm struck the Swedish IM site Aneboda SE14. However, in comparison to the surrounding forests this site managed rather well and a rough estimate considered 20-30% of the area affected. All trees were only thrown on one hectare area. Otherwise, some trees had fallen but others were still standing (Figure 6.7.2). One obvious central opening path showed higher frequency of fallen trees. Estimations to quantify the impact have been carried out.

The storm caused comprehensive damages over a large part of southern Sweden. The natural forest at the SE14 IM site survived fairly well even though some damage was caused. The extent of damage was investigated during the summer 2006. To do this the 50\*50 m plots for vegetation and forest inventory were used (Figure 6.7.3). A number of 47 plots were randomly selected and the plots studied included two circle areas. The large circle was used for trees and the smaller for ground vegetation. The damage was classified 'strong' when more than half of the area was covered by trees and stump turnover occurred. In the weak class, only single trees were felled. In the lowest class no damage could be observed.

The result of this inventory showed strong influence on over 50% of the plots while c. 25% was unaffected leaving 25% in the weak class (Table 6.7.4). Also the forestry board made damage classifications and used air photos showing similar results as the plot inventory.



Figure 6.7.2 Windthrown trees in the Aneboda IM site SE14.

Table 6.7.4 Share of plots in the three damage classes for the IM site SE14, Aneboda affected by the heavy storm in January 2005.

Damage	Plot rate, %	Plot rate, %
	Tree plot	Ground vegetation plot
Strong	53%	51%
Weak	23%	23%
Unaffected	23%	26%

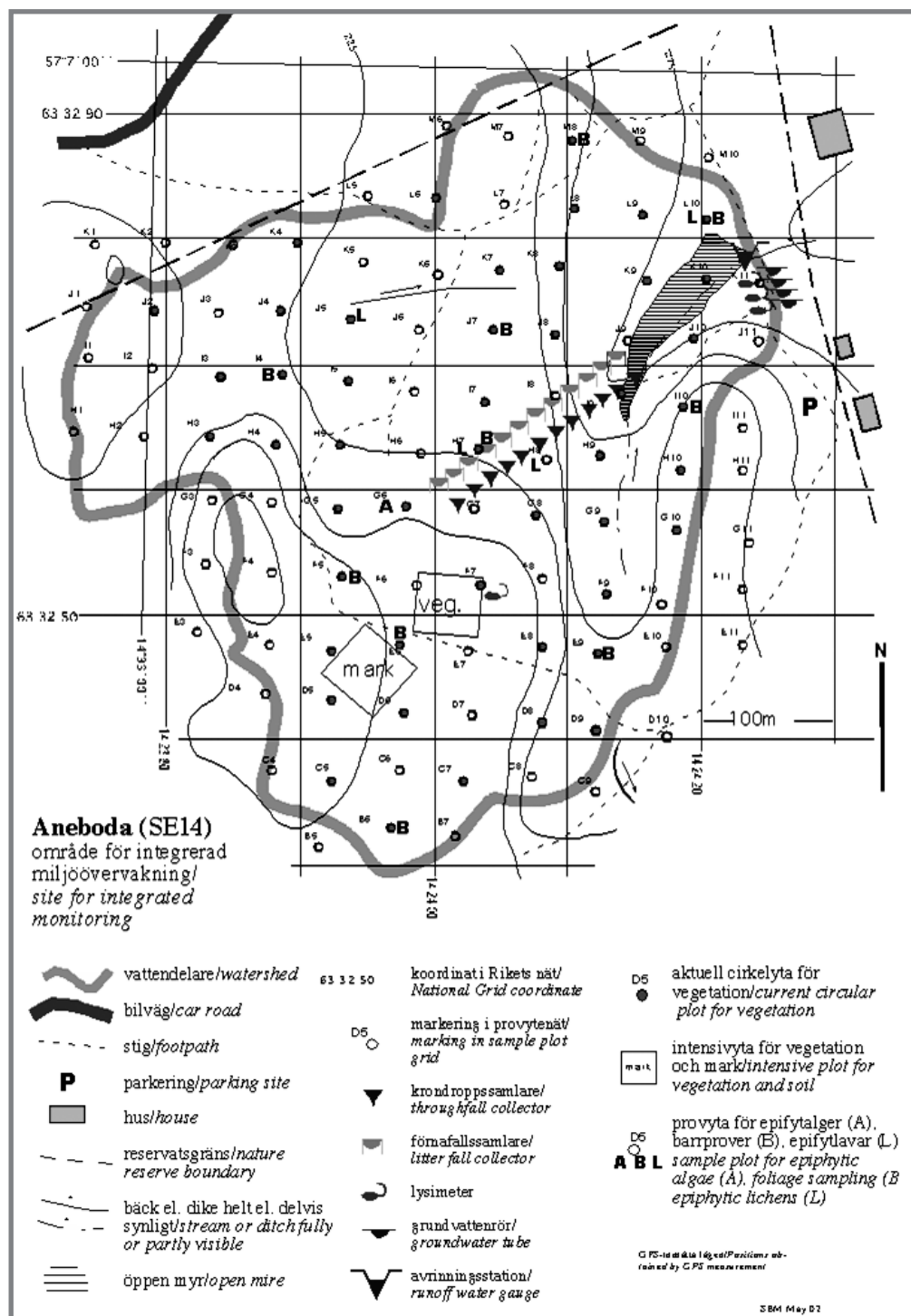


Figure 6.7.3 The IM site SE14 Aneboda with coordinates and 50\*50 m plots where damage investigations were carried out.

## DOCUMENTATION PAGE

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Sammandrag	<p>Programmet för Integrerad övervakning av miljötillståndet (ICP IM) är en del av monitorings-strategin under UNECE:s luftvårdskonvention (LRTAP). Syftet med ICP IM är att utvärdera komplexa miljöförändringar på avrinningsområden.</p> <p>Rapporten sammanfattar de utvärderingar som gjorts av ICP IM Programme Centre och de samarbetande instituten under programåret 2006/2007. Rapporten innehåller:</p> <ul style="list-style-type: none"> <li>● Sammandrag av programmets nuvarande omfattning och IM databasens innehåll</li> <li>● Sammandrag av en rapport sammanställd för utvärdering av miljöeffekterna av Göteborgsprotokollet</li> <li>● Sammanfattningar beträffande utvärderingar inom följande sektorer:               <ul style="list-style-type: none"> <li>○ massbalansberäkningar och kritiska belastningsgränser för tungmetaller på ICP IM områden</li> <li>○ dynamisk modellering av minskningsmål (target loads) för S och N deposition samt inverkan av klimatförändringar på dessa minskningsmål.</li> <li>○ vegetationsdynamik på ICP IM områden i Lettland</li> </ul> </li> <li>● Beskrivning av nationella ICP IM aktiviteter.</li> </ul>			
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